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Design of Studies for Development of Bonneville Power Administration Fish and Wildlife Mitigation Accounting Policy

Phase II

Volume 2: Technical Reports

Final Report 1988

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DESIGN OF STUDIES FOR DEVELOPMENT OF BONNEVILLE POWER ADMINISTRATION FISH AND WILDLIFE MITIGATION ACCOUNTING POLICY

Phase II Final Report

Volume 2: Technical Reports

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INTRODUCTION

Congressional passage of the Pacific Northwest Electric Power Planning and Conservation Act (Regional Act) in 1980 ushered in a new era in natural resource conservation in the Pacific Northwest, A significant feature of the Regional Act was that it established a unique interstate compact, commonly called the Northwest Power Planning Council (Council). Appointed by the Governors of their respective states, two members from each northwestern state--Oregon, Washington, Idaho, and Montana--compose the Council. The Council is charged with developing programs for (1) regional power planning, (2) electricity conservation, and (3) mitigating the effects of hydropower development and operation on fish and wildlife in the Columbia River Basin.

While the responsibility for power, conservation, and mitigation program planning lies with the Council, the responsibility for implementing many of the program measures lies with the Bonneville Power Administration (BPA) and other federal agencies with hydro or power responsibilities in the region: the Corps of Engineers, the Bureau of Reclamation, and the Federal Energy Regulatory Commission. Under the terms of the Regional Act, BPA is required to use its funding authorities to support measures designed, "to protect, mitigate, and enhance fish and wildlife to the extent affected by the development and operation of any hydroelectric project of the Columbia River and its tributaries" (Sec. 4(h)(lO)(A)). Through this mechanism, the costs of mitigating federal hydroelectric development and operation within the Columbia River Basin are to be borne by electrical consumers which purchase power from BPA.

In the years since the passage of the Regional Act, the BPA, the Council, and numerous national and regional agencies, both public and private, have mounted an impressive collaborative effort to protect and enhance the fish and wildlife of the Columbia River Basin and to mitigate damages caused by hydroelectric development and operation. Indeed, estimated BPA costs in this endeavor (including direct expenditures, foregone power revenues, and repayments to the U.S. treasury on behalf of other federal agencies) totaled about $375 million for the period 1983 to 1986. Many of the mitigation measures enacted thus far have proceeded without the benefit of much formal analysis, but in most instances informed judgment has established that such measures are justified.

As the mitigation prescribed by the Regional Act proceeds, the incremental costs of corrective measures to lessen the environmental impacts of the hydroelectric system are expected to increase and difficult questions to arise about the costs, effectiveness, and justification of alternative measures and their systemwide implications. It was deemed prudent by the BPA to anticipate this situation by launching a forward-looking research program aimed at providing methodological tools and data suitable for estimating the productivity and cost implications of mitigation alternatives in a timely manner with state-of-the-art accuracy. In this spirit, Resources for the Future (RFF) agreed at the request of the BPA to develop a research program which would provide an analytical system designed to assist the BPA Administrator and other interested and responsible parties in evaluating the ecological and economic aspects of alternative protection, enhancement, and mitigation measures.

Historical Background

The events leading up to the fish and wildlife provisions of the Regional Act began in the middle and late 1930s when several large dams and powerhouses were created on the main stem of the Columbia River, partly for the purpose of providing employment and other economic stimuli during the Great Depression. The first major dam, Rock Island, a Public Utilities District dam, was completed in 1933, and the much larger federal Bonneville and Grand Coulee Dams in 1938 and 1941, respectively. Toward the end of this early history of hydro development on the Columbia River, it became apparent that an agency would be needed to transmit and market the large amounts of hydroelectricity that would soon become available. To fulfill this need, the Congress passed the Bonneville Project Act in 1937 which created the “temporary” Bonneville Power Administration. Fortuitously, a large market for electricity developed quickly, primal-ily in the electrometallurgical industries that produced aluminum for aircraft construction during World War II. After the war electrical demand in the Pacific Northwest grew steadily and fast until very recently, and hydropower development occurred simultaneously on a very large scale.

The system of federal dams in the region came to be known as the Federal Columbia River Power System (FCRPS). At present it consists of 31 projects with total installed capacity of 19,350 megawatts and over 20 million acre-feet of storage capacity. In addition, there are large public and private utility hydroelectric dams and federal and state dams for flood control. The provided map illustrates the location of major dams within the Columbia River Basin (Figure 1).

Though the FCRPS and other hydroelectric projects provide inexpensive electric power to the region, they also interfere with anadromous fish reproduction and migration. This has led to large losses in potential fish production. But there have been other major sources of such losses, many of which historically preceded hydrosystem development. Logging, mining, agricultural practices and overfishing have hindered anadromous fish production for many years. A large ocean fishery, which developed in time roughly corresponding to the great dam-building era on the Columbia, continues to harvest a large proportion of salmon produced in the Columbia River Basin.

Adding to these factors, a severe drought in the late 1970's and the occurrence of unfavorable ocean conditions reduced fish runs to historical minimums. While the circumstances leading to the passage of the Regional Act stemmed primarily from other sources, the Congress was prompted by environmental concerns, particularly for anadromous fish, to include the following language in the Act:

> 4.(h)(5) The Council shall develop a program on the basis of such recommendations, supporting documents, and views and information obtained through public comments and participation, and consultation with the agencies, tribes, and customers referred to in subparagraph (A) of paragraph (4). The program shall consist of measures to protect, mitigate, and enhance fish and wildlife affected by the development, operation, and management of such facilities while assuring the Pacific Northwest an adequate, efficient, economical, and reliable power supply. Enhancement measures shall be included in the program to the extent such measures are designed to achieve improved protection and mitigation.
>
> 4.(h)(6) The Council shall include in the program measures which it determines, on the basis set forth in paragraph (5), will--(A) complement the existing and future activities of the

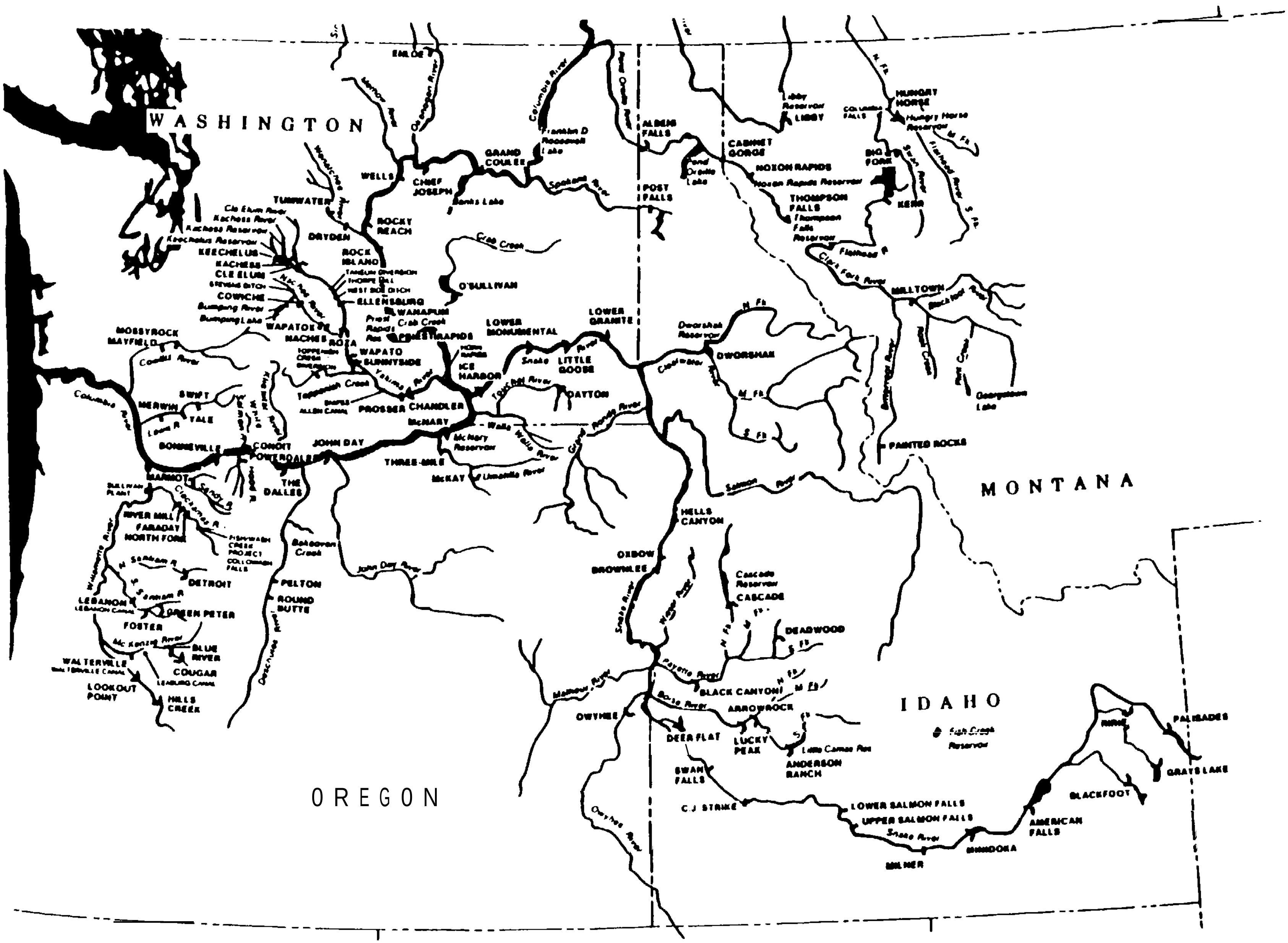

Figure 1. The Columbia River Basin

> Federal and the region's State fish and wildlife agencies and appropriate Indian tribes; (B) be based on, and supported by, the best available scientific knowledge; (C) utilize, where equally effective means of achieving the same sound biological objective exists, the alternative with the minimum economic cost; (D) be consistent with the legal rights of appropriate Indian tribes in the region; and (E) in the case of anadromous fish--(i) provide for improved survival of such fish at hydroelectric facilities located on the Columbia River System; and (ii) provide flows of sufficient quality and quantity between such facilities to improve production, migration, and survival of such fish as necessary to meet sound biological objectives.

Having followed the procedures specified by the Regional Act, the Council adopted its original Fish and Wildlife Program late in 1982. The initial Program contained a variety of mitigation measures, including the installation of bypass facilities to guide migrating young salmon around powerhouse turbines at major dams in the Columbia and Snake Rivers and a special allocation of water for fish, called the water budget. Federal project officers and regulators annually provide the fish and wildlife agencies and the tribes with a total water budget of 4.64 million acre-feet to be used at their discretion between April 15 and June 15 to augment flows normally provided for other purposes, including hydroelectric generation, navigation, and flood control. These enhanced flows, which aid the passage of juvenile fish downstream, are timed in such a way as to maximize their effect. In dryer years when flows fall below average, such as occurred in 1987, providing water budget flows can result in substantial losses in revenue to power producers.

The Fish and Wildlife Program was amended in 1984 and again in 1987 and emphasis remains on the area of the Columbia River Basin upstream from Bonneville Dam. The greatest losses of fish runs have been in the upper Columbia and Snake River areas, while most of the mitigation prior to the Regional Act involved increased hatchery production in the lower basin. The Council has also initiated a process of subbasin planning with priority given to the areas above Bonneville Dam. However, the Council recognizes the need for systemwide integration. Indeed, one of the more important features of the Regional Act is that it specifies that a systemwide approach be taken in the planning and implementation of mitigation efforts.

In the 1987 Program amendments, the Council established an interim objective of doubling the average annual production of adult Pacific salmon and steelhead trout which they presently estimate to be about 2.5 million fish. This total includes fish that are caught at sea and adult fish returning to the mouth of the Columbia River. The Council has not set goals or objectives for specific stocks and subbasins; these products are expected from future planning.

The RFF Research Program

The research program proposed by RFF was intended to be completed in three phases. Phase I, jointly sponsored by the BPA and RFF, was designed to identify economic and related research issues to be pursued in later stages of the research program. A document reporting on Phase I was delivered to BPA in mid-1984. The Phase II research was aimed at providing a comprehensive design of the research program--including development of needed methodologies, identification of data needs and potential sources, and a plan for the program's execution. The bulk of the actual research now contemplated is to be conducted in Phase III, although the research planning has involved considerable research in its own right.

The work plan for Phase II, agreed to by RFF and BPA, specified the following tasks:

Task 1: Investigate the feasibility of, and to propose a plan for development of a system model which would provide capability to estimate loss in fish productivity attributable to development and operation of the hydroelectric system and individual hydroelectric projects and would include the hydrologic, ecologic, and economic components of the Columbia River system, including using suitable (as determined by the contractor) components of existing models.

a) Assess the utility of existing Columbia Basin and Pacific Northwest fish harvest, juvenile migration, and habitat potential models for development of BPA fish and wildlife mitigation accounting procedure and policy. Documentation for- the models . . . will be obtained by the contractor.

b) Prepare a plan for model development which recognizes the need to include components in a system model which would allow simulating the fish production effects of:

i) historic, existing and prospective levels of natural habitat productivity,

ii) alternative harvest management strategies and practice, and

iii) alternative protection, mitigation strategies and practice, and including long-term change in the amount and location of water diversions or instream flow regime, for the purpose of comparing the cost-effectiveness of such alternatives.

Task 2: Design a study to assess alternative procedures for allocating responsibility for loss in fish productivity:

a) to the hydroelectric purpose of federal hydroprojects,

b) between federal and non-federal hydroelectric projects, and

c) to systemwide loss caused by hydroelectric system development and operation, but not attributable to project(s) of any single owner.

Task 3: Inventory available monitoring and accounting options and evaluate their suitability to the objective of formulating a system for measuring mitigation progress, to include:

a) approaches to monitor changes in production of smolts and adult anadromous fish, and

b) study of methods to statistically adjust results of monitoring for random variations and other perturbations to fish production not caused by the hydroelectric system or mitigation efforts.

The Phase II research planning covered all aspects of the work plan but the emphasis placed on various components evolved as the RFF team delved into the nature of the problems to be addressed and as a result of ensuing developments within the region. Specifically, the Council's acceptance in 1986 of an estimate of the loss in fish production attributable to the hydroelectric system lessened the relative importance of developing analytical methods for this task. Most of the effort in Phase II was expended on Task I which (in abbreviated form) called for investigation of the feasibility of, and development of a plan for, a system model including the hydrologic, ecologic, and economic components of the Columbia River system. Where suitable, components of existing models were to be included. The primary motivation for developing such a model (or set of models) is to provide an analytical basis for estimating the biological and economic implications of alternative management strategies.

The necessary steps in developing an analytical system for the Columbia River system follow a natural progression. The first step in developing such a system is understanding the ecological relationships that are inherent within the fisheries. One can then begin to build mathematical models which quantitatively estimate the changes in fish production that might result from management actions. From there, if estimates of the economic costs of alternative management strategies can be made, tradeoffs among levels of fish production and cost can be examined. If the system permits it, advanced analytical techniques may allow one to determine which combination of measures will result in a given level of fish production at least cost.

While this progression from a ecological understanding to cost-effectiveness analyses is straightforward in concept, the complexities of the Columbia River system make the development of analytical methods far from simple in practice. The Phase II final report outlines the technical issues involved in developing an analytical system and proposes a program of research to address these issues. The report is presented in the Summary Report (Volume 1), and the present volume which consists of three technical reports: Part I, Modeling the Salmon and Steelhead Fisheries of the Columbia River Basin; Part II, Models for Cost-Effectiveness Analysis; and Part III, Ocean Fisheries Harvest Management.

PART I

MODELING THE SALMON AND STEELHEAD FISHERIES OF THE COLUMBIA RIVER BASIN

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Part I

Modeling the Salmon and Steelhead Fisheries of the Columbia River Basin

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Cliff Russell contributed to the material presented in Chapter 8. Margaret David and Betsy Lyons assisted in the collection and preparation of background materials.

Chapter 1

SYSTEMS ANALYSIS AND THE FISH AND WILDLIFE PROGRAM

INTRODUCTION

The Fish and Wildlife Program adopted by the Northwest Power Planning Council represents a most remarkable and ambitious collaborative effort to protect, mitigate, and enhance the anadromous fish populations of the Columbia River Basin. The regional scope and immense probable cost of this effort demand that careful consideration be given to program management, i.e., the planning, coordination, and evaluation of measures called for within the Fish and Wildlife Program. The analytical tools which are needed to facilitate program management fall within the purview of systems analysis.

In simple terms, systems analysis can be defined as a body of theory and analytical techniques which are designed to assist policy makers in choosing among options. Two of the more useful tools of systems analysis are modeling, in which complex systems are represented by abstractions, and simulation, a process in which one tries to better understand system behavior and to anticipate potential future impacts of management actions by constructing and experimenting with various computer models. When used correctly, systems analysis can be a vital component of the decision-support system used in natural resource management. The role of systems analysis within the framework of the Fish and Wildlife Program is the focus of this chapter, which begins with a discussion of the general use of modeling to support management and research.

The Use of Models

Modeling and simulation should be an integral part of program management. One view of modeling is as an intermediary between natural resource management and research. Models provide a coherent way of summarizing information gained from past management experience and research, and presenting this knowledge in a usable fashion to resource managers and researchers. For long-term, regional resource allocation problems such as those in the Columbia Basin, it is important that the modeling process keep pace with changes in management philosophy and current understanding of the system. One must understand the dynamic nature of management and research, and the equally dynamic role that models must play.

This view of a dynamic relationship between modeling and components of management and research is depicted in Figure 1.1. Within the system illustrated therein, information is exchanged between components. The only static feature is the management goal. Goals involve implicit values and they are generally stated in ways that make them inherently non-quantifiable. For example, a goal might be "to improve the upriver salmon fisheries." Once defined, the management goal is the primary impetus for management, modeling, and research. The final objective is to have in place measures which serve the management goal effectively. The components other than goal definition receive inputs from, and have explicit feedback loops associated with, one or more additional components. The nature of these components will change with time in response to new or updated information as it becomes available.

In order for this information system to work most effectively, all information pathways shown in the diagram must exist, and information transfer must take place in a timely manner. This is especially true of the feedback loops which pass through monitoring and evaluation and through model corroboration, a systematic process of comparing model structure and predictions to actual system behavior. Premature termination of an information loop can be a invitation to disaster. For example, a tempting shortcut might be to define a management goal, characterize the system involved, formulate a model, run simulations, plan a program of management measures based on model predictions, and implement the chosen measures. Such a strategy may suffice for a localized problem in which the system is reasonably well understood, but it is an imprudent strategy for a large, complex system such as the Columbia. It is unreasonable to assume that one will entirely "get it right the first time." Some measures will work better than expected, some worse, and some not at all. Disappointing of negative results from management actions should not be used simply as

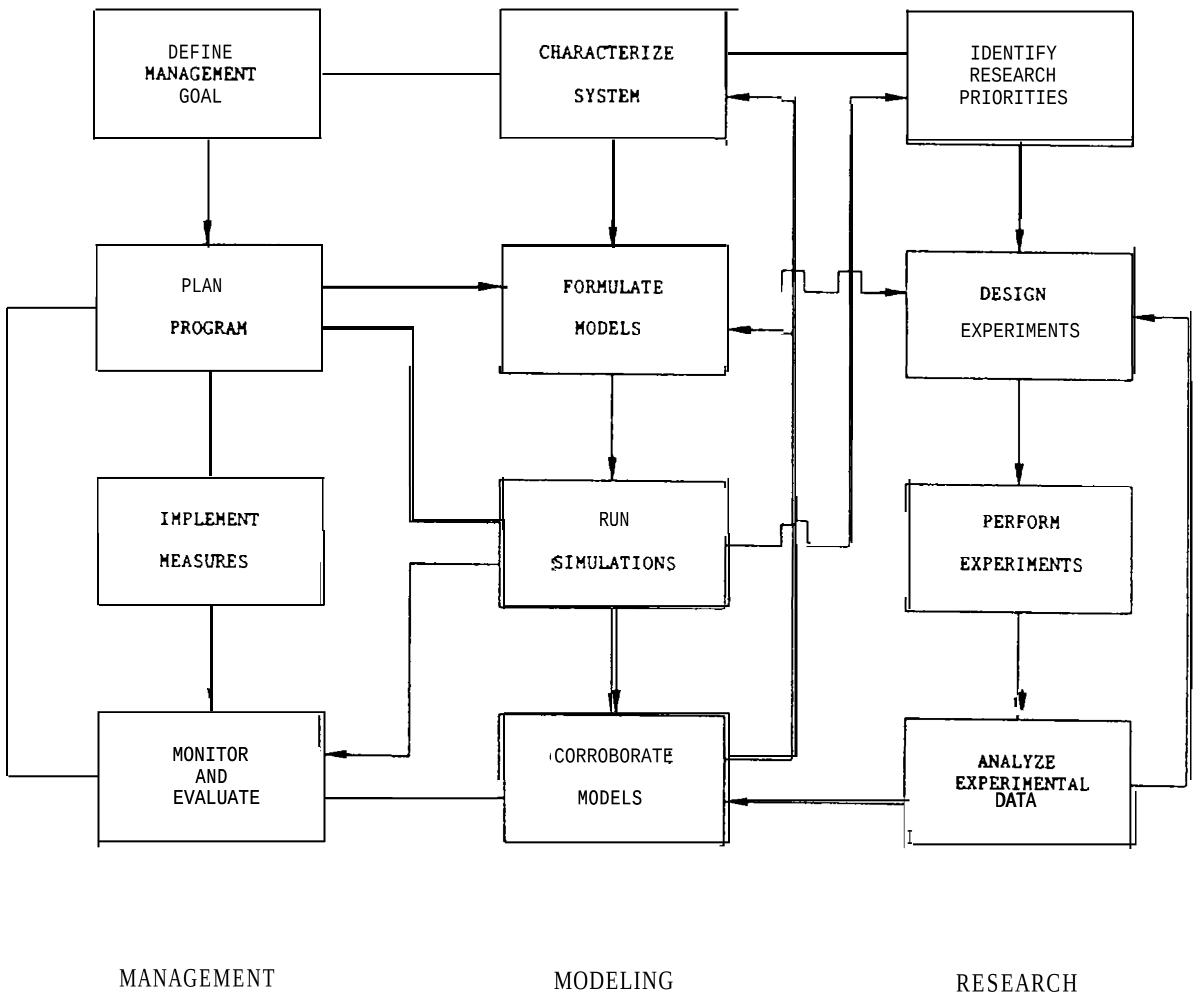


Figure 1.1 Information Flows Among Management, Modeling, and Research.

ammunition for sinking an ongoing modeling program. Rather, knowing that a certain measure performed poorly provides information that should be used to correct inaccurate models which can then be used for future analyses. In a similar fashion, a well-directed research program can be of immense benefit to the management effort by improving the predictive capabilities of the models.

In regard to the Columbia River Basin Fish and Wildlife Program, three areas in which modeling and simulation could play pivotal roles include (1) program planning, (2) system monitoring, and (3) developing a research agenda. Each of these roles are discussed below.

Program Planning

Program planning involves identifying specific objectives which are consistent with management goals, and the measures needed to achieve those objectives. In contrast to goals, objectives are stated such that progress towards an objective can be quantified and measured. For example, an objective might be "to increase hatchery production of chinook smolts by 60% at all hatcheries." The use of systems analysis as a tool for identifying objectives is often overlooked. Frequently, goals and objectives are defined in a political arena far removed from those who then must plan specific measures to achieve each objective.

The unique problems of the Columbia River Basin call for a departure from the conventional approach to defining objectives. Nearly eight years after the passage of the Regional Act in 1980 and five years after the adoption of the initial Fish and Wildlife Program in 1983, specific objectives of the Program still await clear definition. At present, an interim objective of the Fish and Wildlife Program has been broadly defined as doubling the total number (run size) of adult salmon and steelhead trout caught in the ocean or returning to the Columbia River. The delay in defining more specific objectives results from the difficulty associated with defining a set of objectives for the basin that concurrently can meet five basic criteria: biological integrity, internal consistency, equity among all involved parties, consistency with the intent of the Regional Act, and economic realism. As former Council member Kai Lee has noted (personal communication), trying to achieve a reasonable balance among these five criteria makes it difficult to separate objectives from the methods used to achieve them. Perhaps for this reason, the Council has chosen to require that the planning groups which define subbasin objectives must also prescribe the means for achieving them.

While the details of developing system and subbasin plans are deferred to the fish and wildlife agencies and Indian tribes (with appropriate consultation with other interested parties, including BPA), the Council has stipulated general guidelines for the planning process. Two aspects of the Council's recommendations are relevant here. The first is a recognition that "system integration will be necessary to insure consistency." The second is an emphasis on adaptive management as a matter of policy (Northwest Power Planning Council 1987: Section '204). Adaptive management is an approach to reducing uncertainty by viewing management actions as experiments. Careful monitoring of system responses to the management actions provides information about the system which can in turn be used for more efficient management. For a thorough discussion of adaptive management of natural resources, see Walters (1986). Lee and Lawrence (1986) provide a useful discussion of adaptive management in the context of the Columbia River Basin.

Adherence to both a system-wide perspective and the principles of adaptive management necessitates the use of systems analysis. Consistent with a systems approach, the work plan for developing a system plan agreed to by the Council and the tribal and state fisheries management agencies specifically includes the use of the Council's existing System Planning Model as a tool to explore alternative strategies for improving fish runs. Some of the deficiencies in the current System Planning Model for this task are discussed in a later section. The work plan for system planning also identifies several additional tasks, such as systems integration and the examination of production alternatives, that are problematic without the use of models. Models provide a means of checking the compatibility of a proposed set of measures and of identifying mitigation alternatives which might be cost-effective or especially beneficial.

System Monitoring

Efficient management of the renewable natural resources of the Columbia River Basin also requires reliable and accurate monitoring and evaluation of program effectiveness. A primary objective of a monitoring system is to maximize the information gleaned from a fixed amount of monitoring effort. A first step in developing a monitoring system is deciding which measurable components or system attributes should be monitored. Three questions must be considered: (1) which state variables are likely to change in response to management actions, (2) can these changes be measured, and (3) how do these changes relate to overall program success? Models can assist in this process by identifying key variables that are indicative of system behavior. Models can also be used to examine questions of sampling error and to provide insights as to how a monitoring scheme might be structured to reduce uncertainty in parameter estimates. The Council has taken a major step towards ensuring an effective Fish and Wildlife Program by establishing a monitoring and evaluation work group. The importance of models to this task is accented by the Council's decision to place responsibility for maintenance of the System Planning Model with the monitoring work group.

Developing a Research Program

Defining realistic objectives, selecting effective mitigation measures, and designing an efficient monitoring scheme all share a common prerequisite--knowledge of the ecological processes at work within the basin. To this end, a systematic research program is vital to the success of the Fish and Wildlife Program. Such a research program properly should include both basic research and adaptive management actions. Sys terns analysis can assist in the development of a productive research agenda. A major benefit of building simulation models is that models require a formal representation of the system under consideration; one must be specific in defining the relationships between and among components. Major uncertainties soon become glaringly evident.

The Focus of this Report

If models are to be used in the important tasks outlined above, it is in the best interest of all parties concerned that the involved models represent the best available technologies and scientific understanding. This report outlines an approach to modeling the salmon and steelhead fisheries of the Columbia River Basin with special reference to the impact of the hydroelectric system. The term “fishery” is used in the broader sense to refer to the complex of interactions among a fish population, the people which exploit them, and the environment (after Everhart and Youngs 1981: 21).

Our task at Resources for the Future has been to consider the problem of modeling the anadromous fisheries of the Columbia River Basin, design a modeling program which builds upon prior modeling efforts in the region, and develop recommendations for further research. Building explicit models of the Columbia River fisheries was outside the scope of this phase of the research. Therefore, the heuristic models which are presented in following chapters serve only as examples of the technologies which could be employed in a future modeling effort.

SPECIAL CHALLENGES IN MODELING THE COLUMBIA RIVER BASIN

In designing models of the Columbia River Basin fisheries, three major concerns must be addressed. The first of these is the strikingly complex ecological relationships within the system. Within a single stock, there is the spatial and temporal complexity associated with fish that can be spawned in a small stream in Idaho, migrate hundreds of miles down tributary streams and the Columbia River to the ocean, and on to the Gulf of Alaska--and then make a equally impressive (and hazardous) return journey less than five years later. The migratory character of these fish provides ample opportunity for anthropogenic harm through hydroelectric generation, harvest, irrigation, and environmental degradation. This complexity raises two challenges for would-be modelers. The first is trying to understand the system well enough to construct a model. The second is trying to strike a balance such that the models contain sufficient detail to characterize the system but are not so complex as to compromise their utility in planning and policy analysis. Trying to deal with multiple species and stocks such as exist within the Columbia Basin further exacerbates the dilemma.

The second major concern, which is related to the first, is the tremendous amount of uncertainty regarding the system. A long history of fishery research has produced a large literature on the salmonids of the Columbia River Basin. In spite of this literature, important ecological processes or relationships remain poorly understood. For example, little is known about the major processes affecting survival and growth of juvenile salmon in the open ocean, a component which may be crucial in determining the relative success of a stock. Even when processes are reasonably well understood, reliable parameter estimates remain elusive. For example, there is little debate that some fish are killed as they pass downstream through turbines, but how many?

The third major concern relates less to what is known about the system than to what questions might be asked of models. The concern is that models might be asked to address questions that are divergent in scope. To illustrate, two typical questions that might arise are: (1) what is the impact of the water budget on run size; and (2) how will improving bypass facilities at Little Goose Dam affect smolt survival past the dam? The first question is clearly broader in scope than the second and has system-wide connotations. If separate models were designed to address these questions, the model designed to answer the second question would be narrower in scope and have a finer level of resolution. The dilemma is that if only a single model is to be built, then it must have the scope to answer the system-wide question and the resolution to answer the second, more site-specific question. Such a model would likely be so large and cumbersome that it would not be useful to fishery managers.

In light of the concerns identified above, building models of the Columbia River Basin salmon and steelhead fisheries which can contribute to the tasks identified in the preceding section is indeed a major challenge. Addressing these concerns up front can enhance the potential value of models to both fishery managers and hydropower operators in the basin. But even the best-constructed models have little value if used inappropriately.

Models alone cannot resolve the difficulties and uncertainties associated with fisheries management in the Columbia Basin. As Walters (1986: 45) notes, "the value of models in fields like biology has not been to make precise predictions, but rather to provide clear caricatures of nature against which to test and expand experience." The value of models in the Columbia ultimately will depend upon those who use the models.

A PROPOSED MODELING APPROACH

In order to tackle the complex problems posed by the Columbia system, we propose a hierarchical approach to modeling the biological aspects of the salmon and steelhead fisheries. Within this hierarchical structure, models are arranged according to the relative spatial and temporal extent of the system simulated by each model (scope). As the scope progressively increases, the level of resolution within the models decreases. This limits the overall size of the models so that they may fit on a micro- or mini-computer. Separate models of distinct periods in the salmonid life cycle (life stanzas) form the lowest level in the hierarchy, followed by models of the complete life cycle, and at the highest level, a system-level model(s) (Figure 1.2). Individual models within each level of the hierarchy have the capacity to work independently or in tandem. Each model is constructed such that outputs from one model can serve as inputs to other models. Conceptually, the components of one level collectively encompass the next level in the hierarchy. For example, the life stanzas collectively define the complete life cycle of a salmon population, while a collection of populations represent all stocks of interest within a defined system.

The primary reason for modeling life stanzas separately is to allow detailed representation and isolated analysis of each life stanza. Using this approach, one can address questions that vary in scope with a model or arrangement of models that operate at a relatively fine level of resolution. For example, the effect of fallback on upstream survival might be properly examined using a model which simulates only the upstream migration. If increasing the number of smolts passing Bonneville Dam is a key objective, one might analyze alternatives using the juvenile production and downstream migration models with a hydrologic model in a tandem

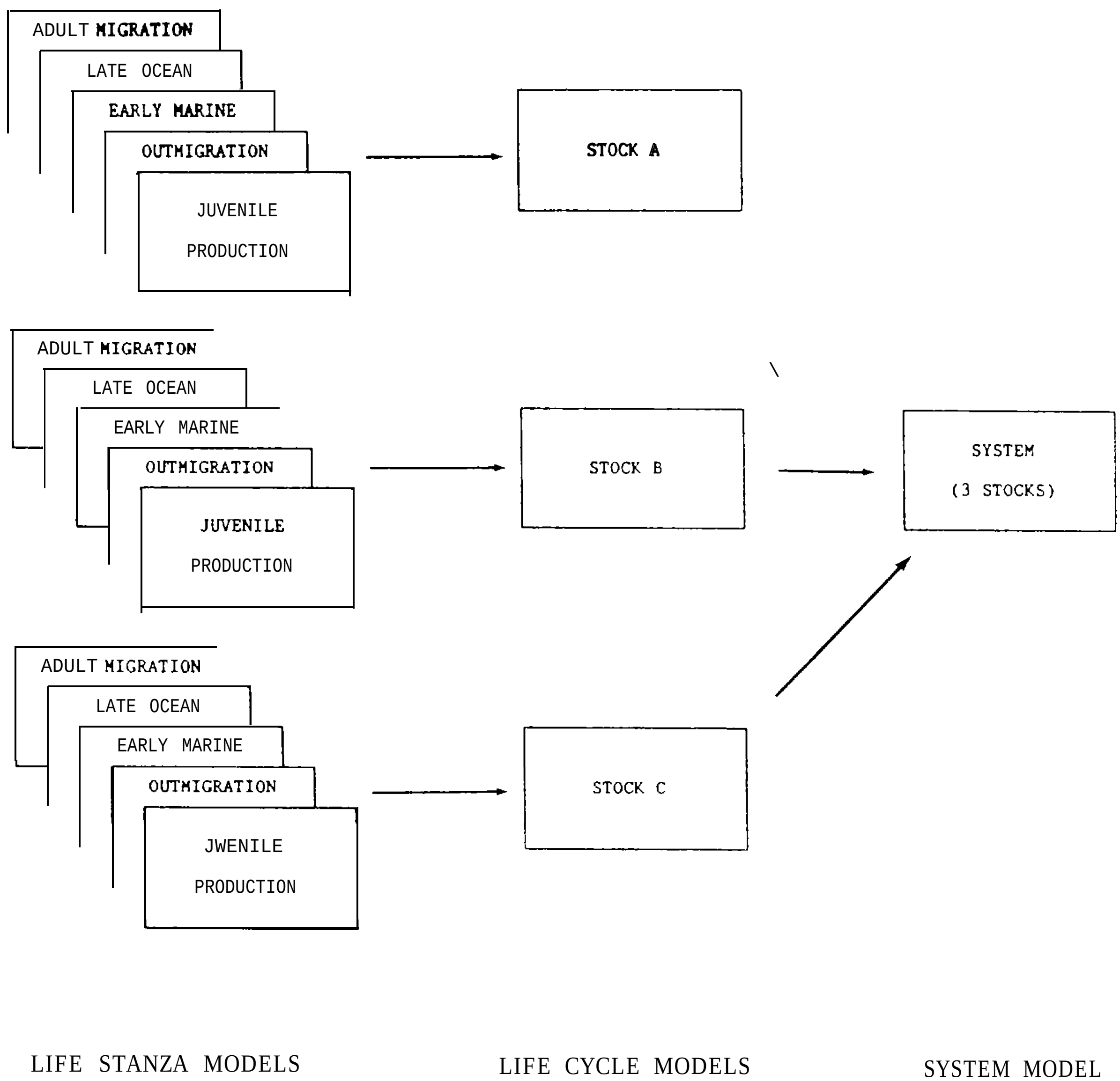


Figure 1.2 Hierarchical Modeling Structure for Hypothetical System Which Includes Three Stocks of Interest. Arrows point in the direction of increasing scope, decreasing resolution.

arrangement. In both examples, the modular structure provides the resolution and the scope necessary to examine the questions at hand without the burden of having to deal with the remainder of the system during the analyses.

Many times questions are asked that concern the complete life cycle. For example, is it feasible to have a sustainable, naturally reproducing population of spring chinook salmon in the Clearwater River Basin? It is possible to address such questions by linking life-stanza models together, but such an arrangement is cumbersome. Also, the level of resolution provided by the life-stanza models is likely unnecessary or inappropriate for population-level analyses. For this reason, life-cycle models are needed for the next level in the hierarchy which operates at a coarser level of resolution. The purpose of a life-cycle model is to simulate the complete life cycle of a particular salmon or steelhead stock. Stock, as used here, refers to a population of fish which is genetically, spatially, or behaviorally distinct from other populations and which shares a common life history among its members. While a plethora of alternative definitions for stock can be found in the literature (see Howell et al. 1985), this characterization is useful from a modeling perspective.

In an analogous fashion, there are system-level questions to be addressed for which life-cycle models are inadequate. The intent of the Regional Act requires a systemwide approach that entails balancing the biological needs of many salmon and steelhead stocks with the often conflicting demands of those harvesting the fish, and of other users of the river system. Because of the temporal and spatial segregation of salmon and steelhead stocks within the basin, certain mitigation measures such as habitat improvement may be relatively stock-specific. Other, more systemwide actions such as the water budget and smolt transportation may benefit a variety of stocks. Both types of measures contribute to the overall goal of the Fish and Wildlife Program by increasing total run size, but a mechanism is needed to evaluate the tradeoffs in terms of costs and equity among systemwide and stock-specific actions. As an illustration, systemwide improvements in downstream passage at first might seem to be prohibitively expensive. However, the costs of passage improvements may compare favorably with the total costs of alternative investments in increasing the juvenile production of upstream stocks on a stock-by-stock basis to achieve similar results. Special concern should be given to actions which may benefit a certain stock(s) while being detrimental to others. For example, increasing hatchery production might lead to excessive harvest pressure on wild stocks, thereby prohibiting these stocks from rebuilding. Addressing questions such as these requires a system model. Such a model does not need the resolution of the life-cycle or life-stanza models, but it does need to faithfully represent the basic ecological relationships inherent in the system.

CONSTRUCTING A HIERARCHY OF MODELS

Conceptually, construction of a hierarchical modeling system might proceed along either of two opposing pathways. One approach is to take an initial, holistic view and begin at the top of the hierarchy with a system-level model. From there, one can progressively divide the system into identifiable components and attempt to elaborate on the mechanisms within each component. In this approach, the behavior of the component models is constrained by, or at least consistent with, model behavior at the next higher level in the hierarchy. An alternative approach to model construction is to start at the bottom of the hierarchy and develop relatively detailed models that are limited in scope. One can then link a number of these models together, observe their joint behavior, and create a higher-level model which mimics the aggregate behavior, but lacks the resolution, of the more detailed models.

The strategy that is envisioned for developing a hierarchy of models for the Columbia River Basin incorporates features of both of the above approaches. Development of a conceptual framework has to begin with a systemwide perspective and follow an approach in which the system is progressively divided. This is necessary in order to insure that a reasonable structure is developed which can address the systemwide concerns of the Council and BPA. It is much easier to put a jigsaw puzzle together if one knows what the final picture should look like. The actual construction of the simulation models may well follow more along the lines of the second approach. It might be instructive and even necessary at times to begin with life-stanza models and use these models to elucidate relationships needed in life-cycle and system models. Development of the life-stanza models is certain to be guided by higher-level considerations. Clearly, any model which hopes to be successful must conform to experience.

A primary requirement of model construction is that logical consistency must be maintained across all levels of the hierarchy. In other words, the behavior of a model in any given level should be compatible with the behavior of the more detailed models at all lower levels, and also compatible with any models which exist at a higher level. If, for example, analyses performed using the downstream migration model (a life-stanza model) indicate a curvilinear relationship between downstream survival and river flow, this curvilinear relationship should be incorporated in the life-cycle and system models. It would be inconsistent to treat survival as being independent of flow in the upper-level models and dependent on flow in the life-stanza models.

Note that the boundaries of the proposed hierarchy are arbitrary. One can extend the hierarchy in one direction to include further divisions of the life stanzas or in the opposite direction to include other major river systems of the North Pacific. At the present time, the boundaries implied by the three-level hierarchy are likely sufficient to deal with the major issues facing the region. However, at some point in the future there may be justification for extending the hierarchy, say to examine downstream survival at a particular dam or reservoir, or to include the Columbia River fisheries in a larger analysis of North American salmon fisheries for purposes of international fisheries regulation. Logical consistency is again the key element to maintain when extending the hierarchy.

RELEVANCE TO THE POWER PLANNING COUNCIL'S MODELING EFFORT

A legitimate concern, expressed by some in the Pacific Northwest, is the compatibility of RFF's modeling effort with that directed by the Council. From its inception, the effort expended at RFF has been designed to complement, rather than duplicate, the modeling work completed under the direction of the Council. Our tactic has been to concentrate on areas where earlier efforts were perceived to be being weak or lacking (e.g., reservoir mortality, estuary and early ocean survival and growth); to place less emphasis on areas which have received considerable prior attention (e.g., downstream passage mortality at dams, juvenile production); and to build on the modeling efforts of the Council and others.

The differences between the models which are proposed in this report and the current System Planning Model (SPM) being used by the Council result principally from the circumstances under which each approach was developed, and the intended use of the models. The model from which the SPM was developed was designed in a two-part, five-day workshop on adaptive management and the Columbia River Basin (see Webb et al. 1986). This workshop served to introduce participants to the concept of adaptive management, and for many as an introduction to modeling as well. The SPM has proven to be a useful tool for organizing information and in providing a systematic way of hypothesizing the relative role of factors affecting fish production within and among subbasins, depending on the location of the subbasins. (Northwest Power Planning Council Staff 1986, Monitoring and Evaluation Group 1988). As previously noted, a more extensive role is planned for the SPM in ongoing subbasin and system planning. Given the expanded role envisioned for this model, it is important to look critically at the current capabilities of the SPM relative to the expectations being raised for its use.

The System Planning Model

Conceptually, the SPM is a simple representation of the salmonid life cycle which tracks fish stocks (in terms of numbers of fish) through time and provides an accounting of the various sources of mortality and production. The lone density-dependent relationship in the model is in the fry-to-smolt survival stage. For this component, the number of non-hatchery smolts of each stock produced increases asymptotically as the number of fry increases. Survival from the smolt stage through adulthood is simply the product of a multiplicative series of survival coefficients, weighted according to passage and harvest parameters specified by the model user. The number of fry produced is a similar weighted, linear function of escapement numbers.

The structure of the SPM leads to predictable dynamic behavior for wild stocks comparable to that of the much simpler population dynamics model first noted by Beverton and Holt (1957). When survival and reproduction parameters are held constant in the Beverton and Holt model, population numbers reach a stable equilibrium size in which the number of smolts produced and the number of spawners remains constant over time. Graphically, the equilibrium size corresponds to the intersection of the smolt-spawner relationship and a straight line with slope equal to the reciprocal of the survival rate from smolt to adult. Management actions or environmental changes which alter either the smolt-spawner relationship or smolt-to-adult survival cause the equilibrium size to change as the point of intersection shifts. In the example shown in Figure 1.3, an increase in smolt-to-adult survival results in a new, larger equilibrium size.

While lacking the succinct nature of the graphic model of Beverton and Holt, the SPM predicts similar results from changes in survival and reproduction parameters when wild stocks are examined in isolation. Under constant conditions, simulated population reach a stable equilibrium level over time. The advantage that the SPM has over the simpler model is that it is possible to keep track of the multiple factors affecting survival and reproduction, including the effects of multiple age classes and interactions between wild and hatchery stocks.

Limitations of the System Planning Model

In the time since the initial version of the SPM was developed, considerable effort has been spent improving versions of the computer model to make the model more accessible to users. A positive result of this effort is that the model is very easy to use and quite accessible. But in the process of making the model easier to use, some of the conceptual flaws of the original model have become more solidly entrenched. Alterations in functional relationships within the model, other than parameter changes, cannot be made without considerable difficulty. This limits one's ability to refine and adapt the model as new information becomes available, or to evaluate alternative hypotheses concerning system processes. Since the internal workings of the model are not apparent to users or adequately

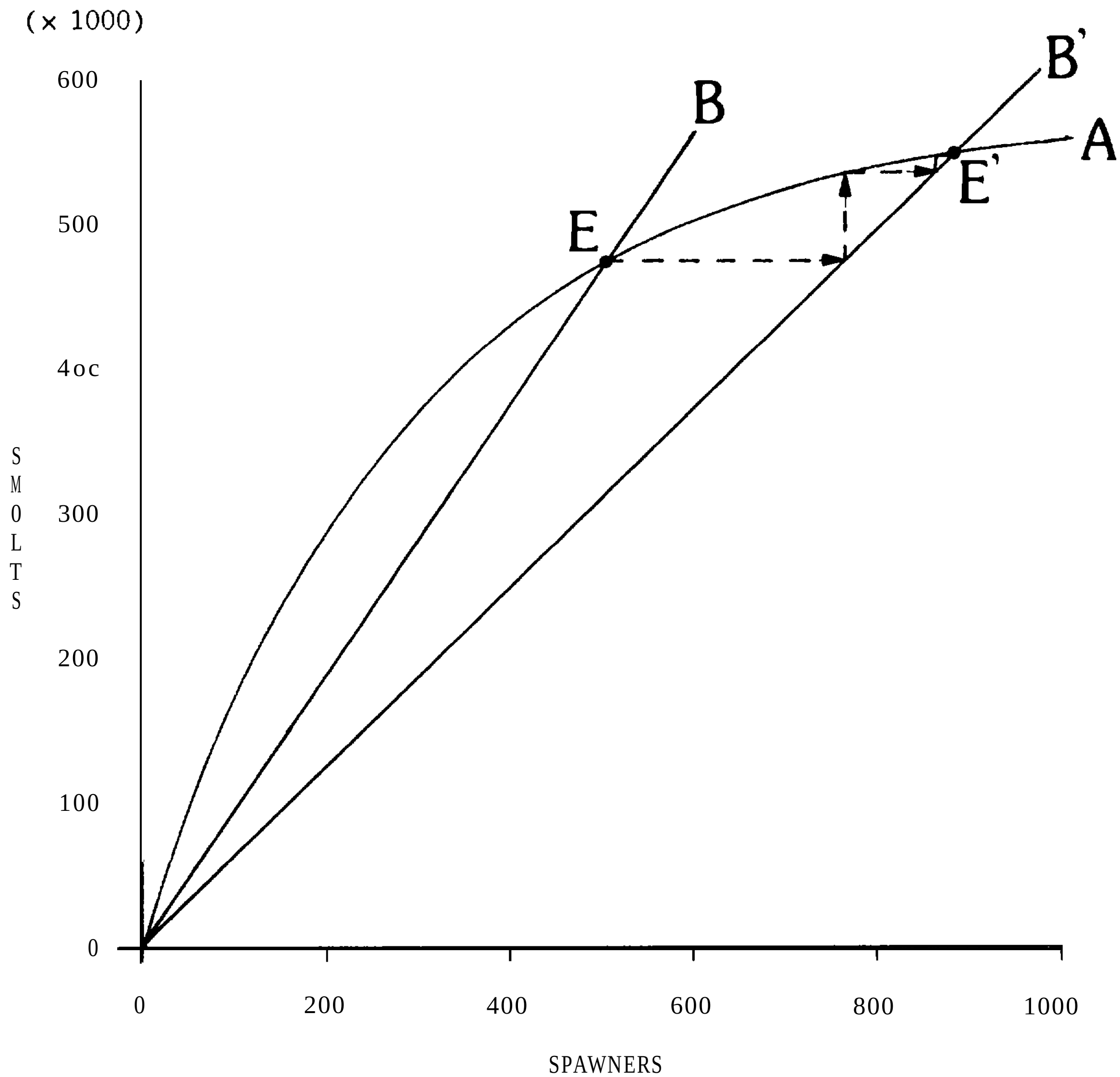


Figure 1.3 Graphic Model of Population Dynamics. Line A represents the number of smolts produced as a function of the number of spawners. Lines B and B′ represent the subsequent number of spawners which result from each level of smolt production for two levels of smolt-to-adult survival. Following an increase in survival rate (B to B′), the population will follow the time path traced by the arrows from point E to the new equilibrium level, point E'.

documented, the chances for misapplication of the model by novice users are high.

One expects to find flaws with any simulation model and it is unreasonable to expect the SPM to be perfect. Many minor problems with the model can be overcome by creative parameterization. But there are several fundamental problems which are more troublesome. While these problems should not affect the value of the model as an organizational or educational tool, they do limit the utility of this model as a primary analytical device for evaluating various protection, mitigation, or enhancement alternatives. Three such limitations are presented by way of example. The first of these is the reliance on a single density-dependent relationship as noted above. In the current configuration, the entire compensatory capacity of a naturally-reproducing population is embodied in the asymptotic fry-to-smolt survival. The rather predictable dynamics of the model are at odds with much of the existing literature on salmonid recruitment. At the very least one should be able to postulate a dome-shaped recruitment relationship (e.g., Ricker 1954) for certain stocks. Peterman (1987) and others have even suggested that the entire class of single-equilibria models may be inappropriate for some salmon stocks.

The second major limitation of the SPM is the manner in which it treats downstream passage survival. In the model, factors determining survival through reservoirs or past projects (with the exception of the proportion of fish which are diverted from the turbine intakes) are independent of biological considerations. When performing a multiple stock analysis, fish size, species, stock composition, and time of migration do not affect mortality rates through spill, turbines, transport, or reservoirs--a clear violation of conventional wisdom. Even more disturbing is the method by which reservoir survival is calculated. The reservoir survival rate per mile (R) is expressed as a piece-wise linear function of flow (Figure 1.4). Subsequently, the total survival rate through the reservoir is calculated as R raised to the Lth power, where L is the length of the reservoir in miles. This results in a most unlikely relationship between flow and total survival through all reservoirs, especially for upriver stocks which commonly migrate 350 miles or more. Figure 1.5 depicts the expected survival through 350 miles of reservoirs as a function

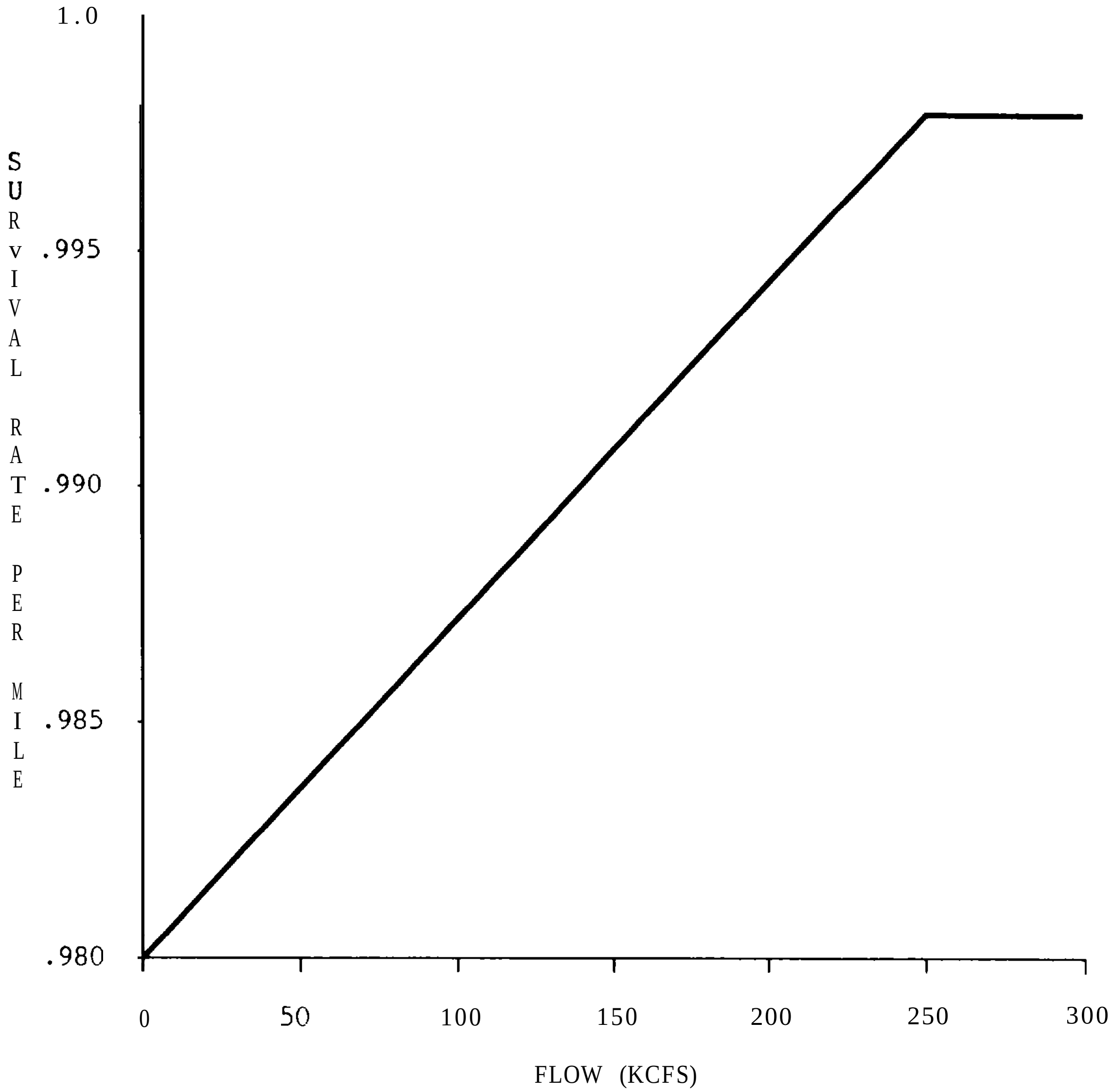


Figure 1.4 Example of the Relationship Between Reservoir Survival Rate Per Mile of Reservoir and Flow as Defined in the System Planning Model.

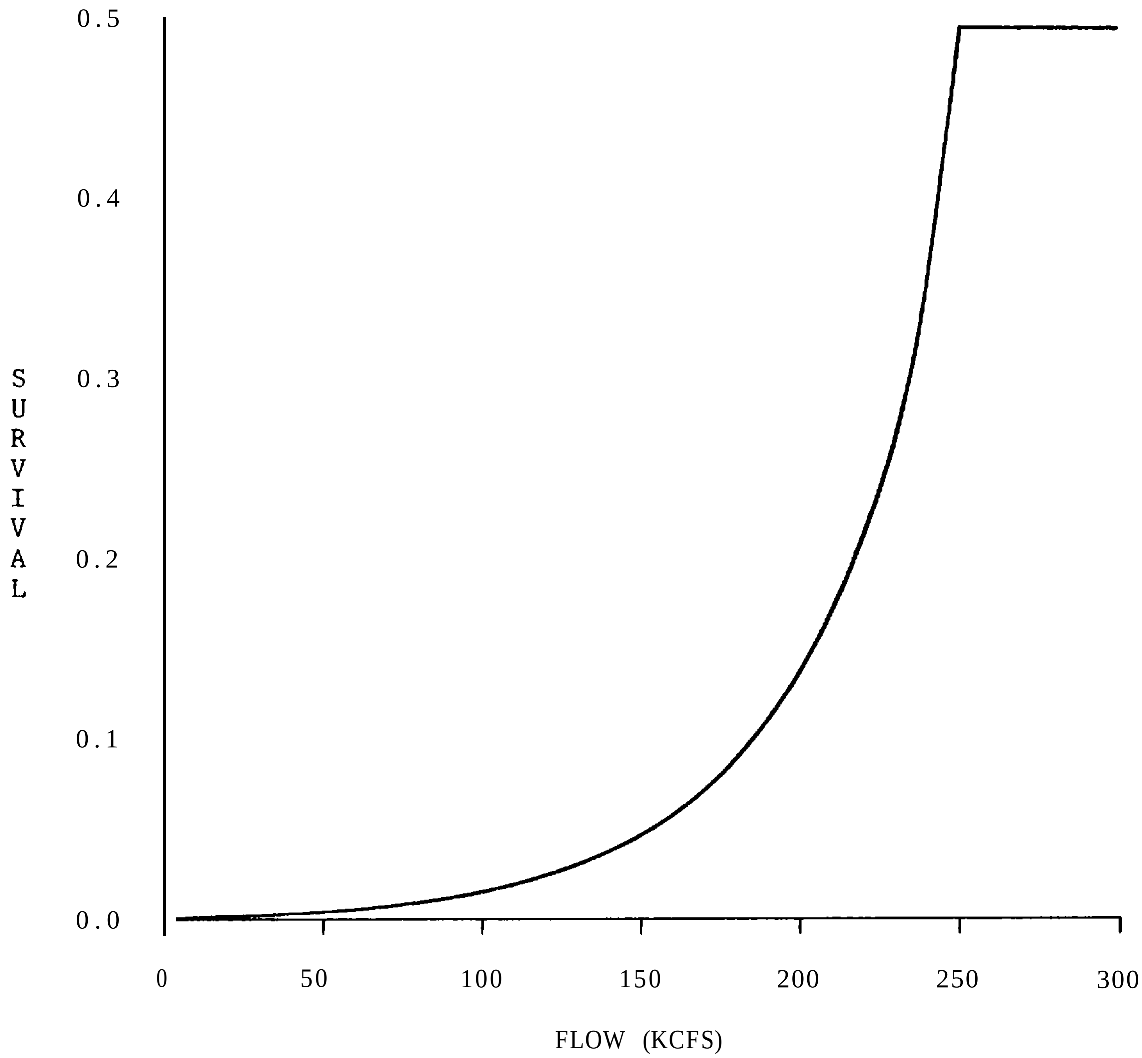


Figure 1.5 Reservoir Survival for Fish Migrating 350 Miles, Expressed as a Function of Flow. The survival rate per mile used in the above diagram equals that shown in Figure 1.4

of flow, using the survival rate per mile values shown in Figure 1.4. From Figure 1.5, a 50 thousand cubic foot per second (KCFS) increase in flow, from 200 to 250 KCFS, results in a 254% increase in reservoir survival, while an equivalent increase in flow, from 250 to 300 KCFS, has no impact on survival. One can imagine the heated debates that would likely ensue between fishery managers and power producers over what constitutes the breakpoint level if water allocation decisions were to be made based on this model.

A third limitation of the current SPM concerns its ability to evaluate specific management alternatives. As presently structured, the model does not provide direct linkages between management actions and model parameters, nor does it provide any estimates of cost associated with such changes. Thus, before one can evaluate the impact of a specific management action, the effect of such action on model parameters must be assessed. This suggests that there must be supplemental models to the SPM to provide the necessary linkage. If cost is to be a part of the analysis, then there must be ancillary economic cost models as well. The final outcome of a comparison will depend as much on these models as on the SPM. But who is going to be responsible for building the supplemental models? Will the Council staff or a committee convened by the Council build such models for each new proposal that comes up? Or will anyone who makes a proposed change in system or subbasin plans be responsible for providing a rationale for altering the model parameters in a certain way? It seems likely that there will be a great deal of confusion and debate among interested parties when the time arises for actually choosing among alternatives.

Differences in the RFF Approach

Our response to the problems raised above is that modeling should proceed deliberately, with the full range of issues incorporated into the modeling process from the start. In contrast to the rather short period in which the SPM was developed, the modeling effort proposed here requires several years to fully mature. The longer time scale is imperative for a comprehensive and detailed examination of the factors influencing fish production, and of the ecological and social consequences of the Fish and Wildlife Program.

Most of the material presented in this report concerns concepts in modeling the fisheries of the Columbia that are not explicitly addressed in the SPM or any other existing model. The central distinguishing feature of the approach presented here is the expanded scope and improved resolution offered by a hierarchical suite of models versus a single model. Specifically, the approach suggested here differs substantially from (and supplements) earlier approaches in the following ways:

- The relatively fine temporal and spatial resolution of the life-stanza models should allow a closer inspection of potential management impacts than do most existing models (the FISHPASS model being a notable exception).

- By integrating information from lower-level analyses, the system-level hierarchical model should facilitate basin-wide analyses that are not currently possible.

- The proposed models include explicit representation of intrastock heterogeneity , a key ecological property.

  Increased reliance on nonlinear and probabilistic relationships within the proposed approach provides a rich exposition of management-fishery relationships.

- Models are to be developed such that calculation of costs are made possible.

The narrowest differences may be between the current SPM and the proposed life-cycle models. Since both are designed to simulate the life cycle of individual stocks, the SPM might be viewed as an excellent prototype life-cycle model. One might expect to modify the internal workings of the SPM to make it more compatible with the overall design, but many of the desirable features of the SPM would be maintained.

Chapter 2

OVERVIEW OF MODELS

INTRODUCTION

The purpose of this chapter is to present an overview of the types of models needed to compose the hierarchy described in Chapter 1 and the application of these models. Most of the preliminary ecological modeling work that has been completed at RFF during this phase of the research effort has focused on the life-stanza models. There are two reasons for this emphasis. First, it is within the life-stanza models that causal links between mitigation measures and biological responses must be specified at a relatively fine level of resolution. This requires a close inspection of the biological impacts of management actions on fish which goes beyond demonstrating that an empirical relationship exists between variables. Since the conditions under which such a relationship was produced may change, it is also important to understand how an empirical result might arise as a function of physical and biological processes. Only by understanding causal relationships can one hope to plan an efficient Fish and Wildlife Program. The second reason for focusing on the life-stanza models is a perceived need to examine particular components of the life cycle which have not received the attention they deserve. Opportunities for enhancing fish production in the Columbia Basin at a minimal cost or effort may be lost simply because they escape consideration.

LIFE-STANZA MODELS

Life-stanza models compose the most basic simulation units in the proposed hierarchical modeling structure. At the lowest level in the hierarchy, each model simulates one of five periods identified in the life cycle of anadromous salmonids: juvenile production, downstream migration, the estuary and early ocean period, the late ocean period, and upstream migration. The output from each model is consistent with the input requirements of the model of the next stage in the life cycle, which facilitates sequential transfer of information. Information sharing among non-adjacent life stanzas also may be necessary, such as a transfer of information between the juvenile-production and estuary-and-early-ocean models.

One of the advantages of having separate models for each life stanza is that unique model structures for each model can be defined using the modeling techniques which seem most appropriate. In the example models provided in subsequent chapters, a broad range of modeling techniques or approaches (e.g., stochastic compartment models, difference equations, and Monte Carlo simulation) are used among the different life-stanza models. These differences in model structure reflect the unique conceptual and physical dimensions which characterize each component. At the same time, having different internal structures does not diminish the compatibility of model inputs and outputs.

The juvenile-production, downstream-migration, and upstream-migration models also may utilize input from a hydrological simulation model, in addition to information required from other biological models. One role of the hydrological simulation model discussed in Part II of this volume is to generate regulated flow information, at an appropriate level of resolution, which might be used by the biological models. The probable scope of a hydrological model includes the Columbia River, its major tributaries, and relevant hydroelectric projects.

In general, construction of life-stanza models requires a broad range of information. The outline in Table 2.1 provides a rough summary of the information necessary to specify each model and the inputs and outputs that might be expected. Considering the range of information necessary to specify all parameters in the life stanza models, existing data sources do not adequately permit stock-specific analyses for all Columbia stocks. Unfortunately, this is true regardless of the level of complexity or resolution of any models which might be constructed for the region. Life-stanza data of any kind simply is unavailable for many stocks. However, this should not hinder the construction of useful models. Our philosophy in developing the models described herein is that modeling should be guided by objectives rather than data. As will be seen in later chapters,

Table 2.1 Outline of Major Features of Proposed Life-Stanza Models.

I. Juvenile Production Model

A. Necessary information
- fecundity relationships
- hatchery production characteristics
- natural production characteristics
- outplanting alternatives
- survival parameters
- growth equations
- smoltification schedules

B. Inputs
- number, sex ratio, age structure, and condition of adults returning to spawning areas

C. outputs
- number, size, physiological condition, and timing of outmigrating juveniles

II. Downstream Migration Model

A. Necessary information
- natural mortality rates
- river flow / migration rate relationships
- dam passage relationships
- transport policies and mortality

B. Inputs
- river flow, hydrosystem operations
- number, size, physiological condition, and timing of juveniles beginning outmigration

C. outputs
- numbers, size, physiological condition, and timing of outmigrants passing each project

III. Estuary and Early Ocean Model

A. Necessary Information
- migration parameters
- growth parameters
- mortality parameters

Table 2.1 Continued.

III. B. Inputs
- environmental conditions
- numbers, size, physiological condition, and timing of of smolts reaching the estuary

C. outputs
- numbers and size distribution of fish recruited to ocean fishery

IV. Late Ocean Model

A. Necessary information
- natural mortality rates
- harvest rates
- maturity schedules

B. Inputs
- numbers and size distribution of fish recruited to ocean fishery

C. outputs
- ocean harvest
- number, sex ratio, age structure, and timing of adults returning to river

V. Upriver Migration Model

A. Necessary information
- natural mortality rates
- harvest rates
- dam mortality rates
- fallback probabilities
- delay time distributions
- energetic cost and reproductive condition information

B. Inputs
- river flow, hydrosystem operations
- number, sex ratio, age structure, and timing of adults returning to river

C. outputs
- inriver harvest
- number, sex ratio age structure, and condition of adults returning to spawning areas

insights gained from simulating selected, or even fictitious, stocks can be valuable in planning a management strategy.

Heterogeneity and Uncertainty

A shared feature among all of the example models is an explicit representation of heterogeneity. In certain instances, means for examining uncertainty also is incorporated in the models. Heterogeneity and uncertainty are important concepts in the context of ecological modeling and the distinction between them should be made clear. In this report, heterogeneity refers to phenotypic or behavioral variation within a population which results from a combination of random environmental factors and genotypic differences among individuals. In contrast, uncertainty denotes a lack of human understanding or knowledge of a process or an event and is associated with the inability to measure or predict. For example, all salmon populations exhibit “heterogeneity” or variance in fish length, i.e., not all fish are the same length. This variation is a population trait which can be described using a statistical distribution with finite parameters. Accurate knowledge of the length distribution depends upon the ability to sample the population. If only a portion of the population is sampled, there will always be a level of “uncertainty” associated with estimates of the length distribution. For example, one might wrongly assume that length follows a specific parametric distribution when an alternative distribution is correct. Even if the assumed distributional form is correct, the parameter estimates may be inaccurate because of some bias in the sampling process, or imprecise because of a small sample size, or both. The difference between the true distribution of fish length and the estimated distribution is never known--hence the uncertainty. A similar analogy could be made regarding uncertainty in relationships among variables.

Since life-stanza models simulate only relatively short periods, changes in the environment which occur over the span of many years might seem to be of minor concern. In reality, there are important ecological linkages between intrapopulation heterogeneity and population responses to both short-term and long-term environmental variation. Intragenerational genetic variation and subsequent differences among individual responses to similar environmental conditions are always present within a population. In the short term, heterogeneity broadens the spatial and temporal pattern of resource use to take advantage of a variable environment, thus decreasing intraspecific competition for limited resources. In the long term, heterogeneity provides an adaptive mechanism for populations to cope with subtle, long-term changes in the environment.

The pervasive nature and ecological significance of heterogeneity are reasons for including it in the modeling process. Holling (1973) remarked that intraspecific heterogeneity, which is often excluded from many model systems, may be a key ecological property which permits real-world species to prosper while their artificial model counterparts, lacking heterogeneity, move toward extinction. In the example models, explicit representation of heterogeneity permits analysis of the relationship between intrapopulation variation and environmental fluctuations. The parameter values defined for a particular model realization assume certain environmental conditions. Environmental changes can be incorporated by exogenously altering parameter values or by defining relationships between parameter values and environmental variables.

One of the purposes for building models is to illuminate information gaps and to provide a means for explicitly including consideration of uncertainty in policy analysis. As mentioned in Chapter 1, the considerable biological uncertainty surrounding the Columbia system compounds the problem of finding an efficient and productive fisheries management strategy. Within some of the examples provided, the life-stanza models have been designed with the flexibility to incorporate alternative hypotheses for unknown or poorly understood relationships. If this approach were to be applied in the Columbia, one could examine the simulated system behavior using alternative relationships and compare the suggested implications. If there are no significant differences in model behavior among alternative representations, one could reasonably assume that the most probable or conventional hypothesis is sufficient. In this case, spending a large amount of money and effort trying to resolve the uncertainty would not be appropriate. Conversely, if model behavior (and by implication, system behavior as well) is found to hinge on a single relationship, efforts to reduce the uncertainty surrounding this relationship through basic research or adaptive management actions would he more justified.

A similar approach can be used to assess the relative importance of uncertainty in model parameters. Once the basic structure of a model has been decided upon, sensitivity analysis can determine which parameters are most influential in determining system behavior and provide some measure of the model's robustness. One of the signs of a good model is that it is fairly robust to small changes in parameter values. If radical changes in system behavior result from minor changes in parameter values, it is wise to question the basic model design.

At times one would like to know what is the best way to proceed, given that several possible outcomes are likely. A set of sophisticated techniques developed by decision theorists holds promise for limited application in such circumstances, particularly in evaluating alternative juvenile production strategies. These techniques often involve clearly defined alternative models and probabilities assigned to the outcomes of each. While the particulars of decision analysis are beyond the scope of this report, Pantell (1976) and Walters (1986) provide the uninitiated with useful overviews of these techniques in the context of resource management. As Walters (1986: 160) notes, building alternative models and assigning probabilities to each in an adaptive management context can "stimulate imaginative thinking about policy options that may be more robust or informative than the options that would otherwise be evaluated."

LIFE-CYCLE MODELS

The purpose of life-cycle models, which occupy the intermediate level in the hierarchy, is to simulate the complete life cycles of salmon and steelhead stocks. While a general model structure may be plausible, modifications in each life-cycle model to account for differences in life histories and geographic ranges among species and stocks will be necessary. The primary reason for having a life-cycle model is so that one may anticipate changes in population structure and size which may result from management actions. However, the emphasis in life-cycle model design should be on the interdependence and interaction among life stanzas--in contrast to the narrower focus of the life-stanza models on the causal relationships between management actions and biology.

The questions to be addressed using life-cycle models are broader in scope than questions examined using the life-stanza models and generally require less resolution. Questions such as the earlier example, "is it feasible to have a sustainable, naturally-reproducing population of spring chinook in the Clearwater River Basin," raise issues of spawning habitat, downstream survival, ocean harvest, upstream passage, inriver harvest, and more. Understanding the proper role of each of these issues in the overall scheme of things is not a trivial exercise. Models that deal with questions of this type need a sufficient level of sophistication to successfully mimic system behavior, but they should not be cluttered with needless detail which only serves to confuse the model user. Finding the proper balance of model complexity and accuracy is a difficult task.

The level of complexity within life-cycle models can be reduced from that of the life-stanza models without substantially losing descriptive capability or logical consistency in a combination of ways. One tactic is to compress a set of related parameters or components into a single, surrogate parameter. For example, egg-to-smolt survival may be decomposed in the juvenile production model into various stages (e.g., egg-to-fry, fry-to-fingerling, fingerling-to-smolt). Such resolution may be unnecessary in the life-cycle model, permitting egg-to-smolt survival to be calculated using a more limited parameter set. A more elaborate means of reducing complexity is to use the life-stanza models to generate empirical relationships between variables. This approach is more appropriate in the case of complex or nonlinear relationships between variables. For example, analysis of estuary survival using the estuary-and-early-ocean model may show a complex relationship between survival and the average length and timing of arrival of smolts entering the estuary. This relationship might be described as a response surface for a given set of environmental conditions where survival is the dependent variable and mean length and time of arrival are the independent variables. Equations which approximate this response surface could be used within the life-cycle model without having to detail the variety of interactions which produce this response.

Since the time frame examined using life-cycle models will cover more than a single year, consideration of interannual variation in the environment could be instructive. While including random effects in models often may seem to serve no other purpose than to generate noise and obscure system behavior, a judicious use of random processes could aid in developing a strategy for evaluating the effectiveness of mitigation measures. If random effects were to be used effectively within a model, they may help indicate the magnitude and nature of a response needed to separate population responses due to mitigation efforts from responses due to random environmental events. Peterman and Bradford (1987) provide an enlightening example of this approach as applied to the English sole (*Parophyrs vetulus*) fishery off the west coast of North America. Their simulation results suggest that under most conditions, the probability of correctly determining time trends in recruitment using conventional methods is extremely low.

In some instances, a model with stochastic parameters might also lead to management implications that contrast with results from a deterministic model. For example, a model which includes a deterministic, monotonically increasing relationship between the number of adult spawners and subsequent migrating juveniles might suggest that improving spawning or rearing habitat is highly cost-effective. A similar model, which includes stochastic elements in the spawner-recruit relationship, might portray habitat improvement as having a much lower expected level of success and suggest that investment in hatcheries or improving downstream migration would be more prudent.

SYSTEM MODELS

The structure of a system model should reflect its primary goal of integrating system components. Having a system model which is comprehensive enough to permit an overall perspective precludes a fine level of resolution if the model is to be of manageable size. Much of what has been said in the previous section about reducing the level of resolution when going from a life-stanza model to a life-cycle model logically can be extended to formulating a system model. Components and parameters must be further lumped together and relationships expressed in simple forms whenever feasible.

It is possible that the structure of a system model may be quite different from the lower-level models. Since a system model will need to simultaneously track many stocks in various life stanzas, some type of matrix or spreadsheet approach may turn out to be the most feasible. In such an approach, the number of individuals within each stock and life stanza might be organized within a two-dimensional “system matrix”, with columns representing stocks and rows representing life stanzas. Each element in the matrix would represent the number of individuals in that specific category during a given time step. The values within the system matrix would change in each time step by multiplying the system matrix by a transition matrix. Coefficients within the transition matrix could be a function of the system matrix values and exogenously determined parameters which reflect system-wide constraints such as ocean harvest rates or dam passage survival rates. Life-cycle and life-stanza analyses could aid in defining the transition matrix coefficients. This type of model structure should be amenable to a variety of commercially available computer spreadsheet software.

Chapter 3

JUVENILE PRODUCTION

INTRODUCTION

A central feature of the Fish and Wildlife Program is the effort to bolster production of juvenile salmon and steelhead in the Columbia River Basin, as measured by both the quantity and quality of outmigrants (smolts). The present natural production of the Columbia system is a small fraction of the estimated production in predevelopment times (Northwest Power Planning Council 1986). Part of this loss can be attributed directly to hydrosystem development which inundated spawning and rearing habitat, increased downstream passage mortality, and blocked passage of adult fish returning to spawn. Other factors such as agriculture, timber harvest, and urban development have also contributed to production losses through environmental degradation. Historically, the dominant means of mitigating production losses has been the use of hatcheries. Present efforts to increase production levels include a combination of both natural and artificial production methods, including using hatcheries to produce fry which are then released (outplanted) in natural streams for rearing. Additional measures which have been implemented or are proposed for the Columbia River Basin include the construction of additional artificial production facilities, water management schemes to provide suitable instream flows for naturally spawning fish, and studies to aid in the detection, diagnosis and control of fish diseases and parasites (see Northwest Power Planning Council 1987: Section 700).

For each enhancement measure that is proposed, the questions arise, "what impact will this measure have on juvenile production; will it be effective?" In reality, the answers are never known *a priori* and often remain unknown even after the measure has been implemented. Production enhancement is fraught with uncertainties and evaluating the success of a measure may be problematic, prohibitively expensive, or simply not considered a priority item. The impact of habitat enhancement on natural production has been especially difficult to predict and evaluate. As Everest et al. (1985: 113) note, "the risk of failure to achieve biological objectives of enhancement is high without a thorough pre- and postproject evaluation." Unfortunately, habitat enhancement measures accompanied by comprehensive research are rare and a thorough evaluation may cost as much as the enhancement measure alone. While it may seem easier to obtain a quantifiable increase in production using artificial means, hatcheries are not without problems. Outbreaks of disease, reduced genetic diversity, and unnatural rearing conditions can contribute to poor survivability of smolts following release, leading to less than anticipated production of adults. Outplanting, which combines features of both artificial and natural production, has its own peculiar set of problems that have not been fully elucidated. Most knowledgeable authors advise extreme caution when proceeding with outplanting (see Nickelson et al. 1986; Reisenbichler and McIntyre 1986; Smith et al. 1985).

The uncertainties associated with juvenile production make it difficult to choose the most effective enhancement strategy from among a set of alternative measures. In an idealized world where unlimited resources are available to devote to enhancement, each proposal might be judged solely on its own merits. Following a careful analysis, those measures which demonstrate an expected net positive impact might be pursued. In the real world where resources are limited, proposed measures cannot be considered in isolation but rather they must be examined relative to available alternatives. Each alternative is judged based upon expectations suggested by some type of model, where in this case “model” refers to an assumed set of quanitative relationships. These models can be simple or complex and include varied levels of uncertainty, depending upon the available information and the nature of the proposed measure. For example, the decision to build a hatchery might be based upon a fairly complex analysis of the suitability of several sites to support the hatchery, and the capacity of the hatchery to address the needs of the fishery. In contrast, a decision to provide increased flows to a section of river might be based on relatively simplistic assumptions about the expected increase in available spawning area that will result from flow augmentation. The dilemma which confronts fisher-y managers is how to choose among alternatives when the information driving the decision process is derived from diverse sources and is of variable (and perhaps unknown) reliability.

CONCEPTUAL MODELING OF JUVENILE PRODUCTION

Trying to predict juvenile production or recruitment is one of the more enduring and perplexing problems facing fishery managers. There is no concept more basic, yet so poorly understood, in fisheries science than juvenile production. The fisheries literature abounds with proposed methods and discussion of inherent problems in trying to predict salmonid production. Envirosphere (1985a, 1985b) provides a useful review of this literature and its applicability to the Columbia River Basin. Despite a large background of research, fishery managers are commonly frustrated in their attempts to reliably forecast the numbers (or biomass) of young fish that will be produced or recruited to a fishery.

An examination of juvenile production from basic principles reveals a complex, multi-dimensional phenomenon. Ignoring for the moment the complexities involved in realizing a given spawning escapement (subsequent chapters will return to this topic), the essence of the salmonid recruitment problem is estimating the level of juvenile production that can be expected from a given number of spawning adults. The process by which spawners give rise to smolts can be represented conceptually as a multiplicative series of coefficients which reflect biological transitions:

$$R = S \cdot f_1 \cdot f_2 \cdot f_3 \cdot s_1 \cdot s_2 \cdot s_3, \qquad (3\text{-}1)$$

where

R = number of smolts produced,

S = spawning escapement,

f_1 = mean number of females per spawning adult,

f_2 = mean number of redds per female,

f_3 = mean number of eggs per redd,

s_1 = egg to fry survival,

s_2 = fry to parr survival,

s_3 = parr to smolt survival.

The steps which reflect egg-to-smolt survival (s_1,s_2,s_3) are defined arbitrarily and could be divided further- to include shorter critical periods such as summer and winter survival of parr. As defined above, Equation 3.1 only applies to natural production; the coefficients could be redefined to represent artificial production as well.

Two characteristics of the coefficients in Equation 3.1 are important. First, all coefficients are influenced by the environment and thus are certain to vary from year to year. Second, the coefficients may not be independent of each other. In the case of density-dependent mechanisms, the value of each coefficient will depend on the values of the preceding coefficients. Letting E_m be a surrogate index of environmental conditions under which the spawning stock reaches maturity, E_s reflect environmental conditions in spawning areas, and E_r reflect environmental conditions in rearing areas, the relationships among coefficients and the environment for a particular stock can be expressed in a relational matrix. An X within the matrix indicates that the coefficients noted by the row headings are functions of the entities noted in the column headings. A vertical dotted line divides environmental factors from population-density factors. The matrix below illustrates this approach for natural production.:

	E_m	E_s	E_r	S	f_1	f_2	f_3	s_1	s_2
f_1	X								
f_2	X	X		X	X				
f_3	X	X				X			
s_1		X		X	X	X	X		
s_2		X	x	x	X	X	X	X	
s_3			x	x	X	X	X	X	X

For example, the average number of redds per female (f_2) is interpreted as being a function of pre-spawning environmental conditions (E_m) which influence the spawning capacity of individual females, the number of spawning females $(S \cdot f_1)$, and the usable spawning habitat (E_s). Similarly, parr-to-smolt survival (f_3) is a function of parr abundance $(S \cdot f_1 \cdot f_2 \cdot f_3 \cdot s_1 \cdot s_2)$ and rearing environment (E_r). Since some of the factors controlling parr abundance are in turn a function of E_m and E_s, parr-to-smolt survival is indirectly a function of maturation and spawning environment as well.

Even within this simple abstraction there is considerable interdependence among factors determining the level of natural production. The dimensionality of the problem is obscured to some extent by the use of the environmental indexes, E_m, E_s, and E_r. Each of these indexes represents the distillation of interrelated, multi-dimensional components. For example, an index of rearing environment must include the physical dimensions of substrate and water quality and quantity in addition to biological factors such as interspecific interactions. Thus, an expanded relational matrix which included all of the physical and biological factors which play a role in natural production would be considerably more complex than the one above.

If one further expands the relational matrix to also include alternative modes of producing smolts within the basin (e.g., hatcheries, outplanting) and the interactions that can occur among production methods, the matrix becomes excessively large and hopelessly complex. Also, as artificial means of production are introduced into a system, questions of relative survivability, genetic diversity, and incidence of disease become more prominent. It is no longer sufficient to consider only the quantity of smolts produced--the quality of the fish must also be addressed.

Fish quality is a nebulous concept that deserves further comment since it has implications in life stanzas beyond juvenile production. Fishery managers and biologists seem to have a variety of definitions for "quality" but basic to each of these is the concept that all fish are not the same (see McIntyre 1987, Warren 1987, and others in Bouck (ed.) 1987). In simple terms, high-quality fish are "better" or more desirable than low-quality fish. Quality is a term that may be applied to individual fish or to populations. When referring to individual smolts, quality generally refers to some measure of the potential of each fish to survive to adulthood. An index of quality might also be applied to adult fish to indicate relative fecundity or fitness. Alternative definitions of quality which apply to populations might include components of genetic diversity--a feature which improves the relative fitness of an entire stock in the face of a variable environment. Regardless of the definition chosen, incorporating fish quality into a modeling framework requires a precise definition of quality in quantifiable terms.

QUANITATIVE MODELING OF PRODUCTION

While it is comparatively easy to construct a relational matrix such as the one above, there are a host of problems associated with translating such a matrix into precise, quantitative models. Two fundamental steps in building production models are (1) deriving meaningful measures or indexes which accurately characterize the environment, and (2) deciding the relative importance of population-density factors relative to environmental factors, i.e., how sensitive are density-dependent relationships to changes in the environment. Neither of the two most common types of juvenile production models, “habitat-based” models and “stock-recruitment” models, adequately address both of these problems.

The problem of quantifying the environment is a product of the multidimensionality alluded to earlier. Biological organisms are wonderfully intricate integrators of their environment. The physical state of a living organism is a function not only of the present environment in which it exists, but also of all past environments that it has experienced. Thus an ideal index of rearing environment, for example, would include components of the spatial and temporal heterogeneity of the mosaic conditions experienced by young fish.

Attempts to develop such indexes or to select indicator variables which can explain significant amounts of variation in smolt production have been made within the habitat-based production models. These models are designed to estimate the potential production of an area, under the assumption that juvenile production is limited by the quantity and quality of the habitat rather than the number of available spawners. Much of the criticism surrounding these models involves this assumption, the extensive amounts of data that are required (for the more sophisticated analyses), and the site-specific nature of the models (see Envirosphere 1985a). The failure of these models to address interspecific interactions may also lead to disappointing results (Li et al. 1983).

In contrast to the habitat-based models, stock-recruitment models have been developed which estimate recruitment as a curvilinear function of parental stock size. The most common of these models are those introduced by Ricker (1954) and Beverton and Holt (1957) and the many variations on these basic models which subsequently have been proposed (see Rothschild 1986). In the stock-recruitment models, survival from the egg stage to age of recruitment (in this case outmigration) decreases as the number of eggs produced increases. While such models may involve only a small number of parameters, fitting of model parameters generally requires relatively long time series of data, and the assumption that environmental factors which affect recruitment have remained constant over time. In practice, stock-recruitment relationships are known for their notoriously poor fits to empirical data due to the large amount of unexplained variation in recruitment; yet these models are widely used as heuristic tools.

APPLICATION OF PRODUCTION MODELS IN THE COLUMBIA RIVER BASIN

The shortcomings of the habitat-based models and of the stock-recruitment models are severe detriments to the singular use of either approach throughout the Columbia system. On the one hand, the habitat-based models are poorly adapted for use in situations where the number of spawning adults is limited and one wishes to anticipate increases in juvenile production which might result from increases in the number of returning adults. Similarly, stock-recruitment models, which are primarily designed to estimate harvestable surplus, are not adept at incorporating changes in the environment. Estimating the effect of habitat changes on stock-recruitment parameters in the absence of a long time series of "before and after" data is highly speculative.

Production models which are needed for the Columbia River Basin need to have components of both habitat-based models and stock-recruitment models. In a region where native stocks are being rebuilt largely on the basis of habitat improvement and improved smolt-to-adult survival, both environmental and density-dependent factors must be considered. In addition, the emphasis on using outplanting as a means of supplementing natural production demands that careful consideration be given to ways of modeling interactions between wild and hatchery fry.

Given the mixed assortment of mitigation measures that have been proposed for tributary streams, the inconsistencies in available data, and the diverse character of the tributary basins, it seems unwise to attempt to develop a general production model that can be applied ubiquitously. A more pragmatic approach might be to focus on the tributary basins and try to develop production models that are peculiar for each basin. Such models would take advantage of the best available data for each basin and be tailored to fit the physical characteristics of each. Available management options within each model would be limited to those measures which are identified beforehand as being appropriate for each basin. To some extent, this approach has been followed in the Columbia and it is expected that the planning teams involved in subbasin planning will utilize methods which seem most appropriate for the basins for which they are responsible.

From a system-wide modeling perspective, the major requirement of subbasin production models is that they produce comparable outputs. These outputs will feed into other models of subsequent components of the system and thus must meet fairly rigid format requirements. The minimum information which should be reported for a given escapement level is the number of smolts produced for each stock, the timing of the outmigration, and perhaps qualitative information such as length distribution of smolts and a relative index of hardiness.

For the policy makers, it would be beneficial if the production models which are developed explicitly incorporate elements of uncertainty as alluded to in Chapter 2. There are a variety of ways to modify or experiment with deterministic models such that they produce a range of outputs with associated probabilities rather than single point estimates. Those familiar with modeling and with an interest in system-wide concepts need to work closely with the subbasin planning groups for the mutual benefit of all.

Chapter 4

MODELING DOWNSTREAM MIGRATION

INTRODUCTION

Each year millions of young salmon and steelhead trout released from hatcheries or reared naturally in tributary streams throughout the Columbia River Basin begin a perilous journey to the Pacific Ocean. Hydroelectric development of the Columbia System has replaced the once free-flowing rivers encountered by these fish with a series of slow-moving reservoirs punctuated by run-of-river dams. The considerable impact of development and operation of these dams and upstream storage reservoirs on outmigrating juveniles (smolts) is well documented (e.g., Bentley and Raymond 1976; Ebel et al. 1979; Raymond 1968, 1969, 1979; Schoeneman et al. 1961) and has prompted an extensive program of mitigation measures (see Northwest Power Planning Council 1987: Section 400).

The most obvious hydrosystem impacts on smolts are those that occur at the dams and powerhouses. When young fish are swept through turbines, those that are not killed outright may be stunned or injured, making them easy prey for predators or disease. Fortunately, not all migrating fish must pass through turbines. Some smolts pass through the spillways as water is spilled. Other migrants, at dams which are suitably equipped, are diverted from turbine intakes by mechanical screens and passed through bypass channels. Diverted fish may be returned to the river immediately downstream of the dam or collected and placed in trucks or barges and transported to below Bonneville Dam, thereby avoiding intervening dams.

Less obvious, but perhaps more insidious than the turbine-related impacts are the indirect losses caused by the reshaping of river flow which accompanies hydroelectric development. The migration of juvenile salmon from their natal freshwater streams to the Pacific Ocean is a remarkable natural phenomenon in which timing plays a central role. The onset of this transition begins with the the parr to smolt transformation (smoltification) which is cued by changing photoperiod and temperature. During this period, the young fish experience physiological changes that enable them to adapt to changing osmotic conditions in their environment. If water temperatures become too high or if smolts are excessively delayed in reaching salt water, the smoltification process may be reversed (Wedemeyer et al. 1980). This results in fish which are ill-conditioned for the marine environment and have a reduced chance of survival. The apex of the smoltification process for most stocks is set to coincide with the onset of peak spring flows, when the Columbia and Snake Rivers are swelled with runoff from the melting snowpack. Historically, the young salmon and steelhead have taken advantage of naturally high flows to catch an easy ride to the ocean. Currently, the huge storage reservoirs scattered throughout the basin severely limit streamflow levels as water is stored for future electrical generation and irrigation. This reduction in flow, combined with reduced momentum and an increase in cross-sectional area of the river due to the run-of-river dams, precipitates a drastic decline in water velocity, thereby slowing the rate of travel of outmigrating smolts.

An end result of hydrosystem development and operation is that smolts migrating from their headwater streams of origin are now faced with a passage that takes longer, is more physically taxing, and includes new new sources of mortality, as compared to pre-development times. Models are needed which can help assess hydrosystem impacts and evaluate the relative effectiveness of potential management actions such as improvments in passage facilities, transportation, and providing increased flow for fish (e.g. the water budget), measures which are designed to improve survival through the system.

Models which consider downstream migration of smolts have been previously developed. Currently, the FISHPASS model developed by the Corps of Engineers (Tanovan 1985) is the most complex of the existing models and is often used by those investigating downstream passage problems in the Columbia River Basin. The purpose of FISHPASS is to provide a detailed accounting of the direct losses incurred in downstream passage. While a fairly rigid structure for accounting for losses incurred at the dams is included, treatment of reservoir (pool) survival is less emphatic. Three contrasting options for calculating reservoir survival are provided. Reservoir survival can be specified as (1) a constant value, independent of streamflow, (2) a monotonic increasing function of streamflow based on consideration of travel time, or (3) a bell-shaped function of streamflow. The Council's system planning model (SPM) includes a downstream migration component which is less sophisticated than FISHPASS but which accounts for dam-passage losses in a similar fashion. Reservoir mortality in the SPM is expressed as a function of streamflow on a per mile basis (see previous discussion pp. 1-18 to l-20). Both FISHPASS and CSPM include options for transporting fish around dams. Neither model explicitly incorporates stochastic processes other than variation in streamflow into the treatment of downstream passage. In contrast to the numerical "accounting" approaches, Rondorf et al. (1985) proposes a bioenergetic approach to modeling seaward migration of smolts as a method of integrating cumulative impacts. Demonstration of the utility of this approach awaits further development of this model.

The variety of approaches to modeling reservoir survival incorporated in existing models highlights the uncertainty surrounding this issue. There is an immediate need for a sound, theoretically and empirically based model of reservoir passage and survival. Modeling experiments performed using FISHPASS and the SPM illustrate the importance of reservoir survival to the overall success of the migration. Continuing to use ad hoc relationships that are based primarily on intuition to depict reservoir passage is unlikely to sufficiently illuminate system processes to the point where mitigation measures such as the water budget can be properly evaluated. Problems still remain l-elated to smolt passage at individual dams, but the research program currently directed at these problems by the Corps of Engineers, BPA, and others seems commensurate with the task. The same cannot be said for the current research program aimed at reservoir mortality and water budget effectiveness despite recent efforts directed by the Council to establish a research plan to examine this issue. As discussed in Chapter 8, the existing information base does not sufficiently support the contentions on which current water management practices (including the water budget and spill) are based. The problem in the past has not been that too little effort has been expended collecting data (the smolt monitoring program of recent years has in fact collected an abundance of data), but rather that it has been seemingly impossible to reconcile much of the data with the inherent assumptions of the analytical framework used for interpretation.

The modeling approach described in the following sections focuses on the processes involved in reservoir passage. Dam-passage and transportation issues are not addressed in this report. The conventional approaches to modeling these components adopted in both FISHPASS and the SPM seem to be adequate at this time. This is not to suggest that either FISHPASS or the SPM are totally accepted by all parties. Considerable disagreement exists on various components of each model. One of the more debated issues involves the manner in which transportation benefits should be credited to dams. This is a purely arbitrary accounting decision which should be made by regional policy makers; it is not a technical issue in modeling. By emphasizing the reservoir passage component, the modeling approach presented here should complement rather than supplant existing models such as FISHPASS.

CONCEPTUALIZATION AND FORMULATION

The downstream migration of smolts is conceptualized as an irreversible particle diffusion process. Smolts pass from an initial source through a sequential series of compartments enroute downstream, with some loss to mortality (Figure 4.1). The source may be a hatchery that is releasing smolts into the river, or a tributary stream from which naturally produced smolts are entering voluntarily. The river is divided into compartments (i.e., reservoirs or reaches) delineated by physical boundaries such as dams or the confluence of a major tributary. Individual smolts do one of three things within a given time interval: (1) remain in their present compartment, (2) pass to the downstream compartment, or (3) are lost via mortality.

The quantities of interest are the number and timing of individuals entering each compartment or completing the migration and entering the estuary. Let T_i be a random variable that measures the time of passage for a given individual from compartment i to compartment i+l. Conditional on an individual having entered compartment i at time zero, the probability

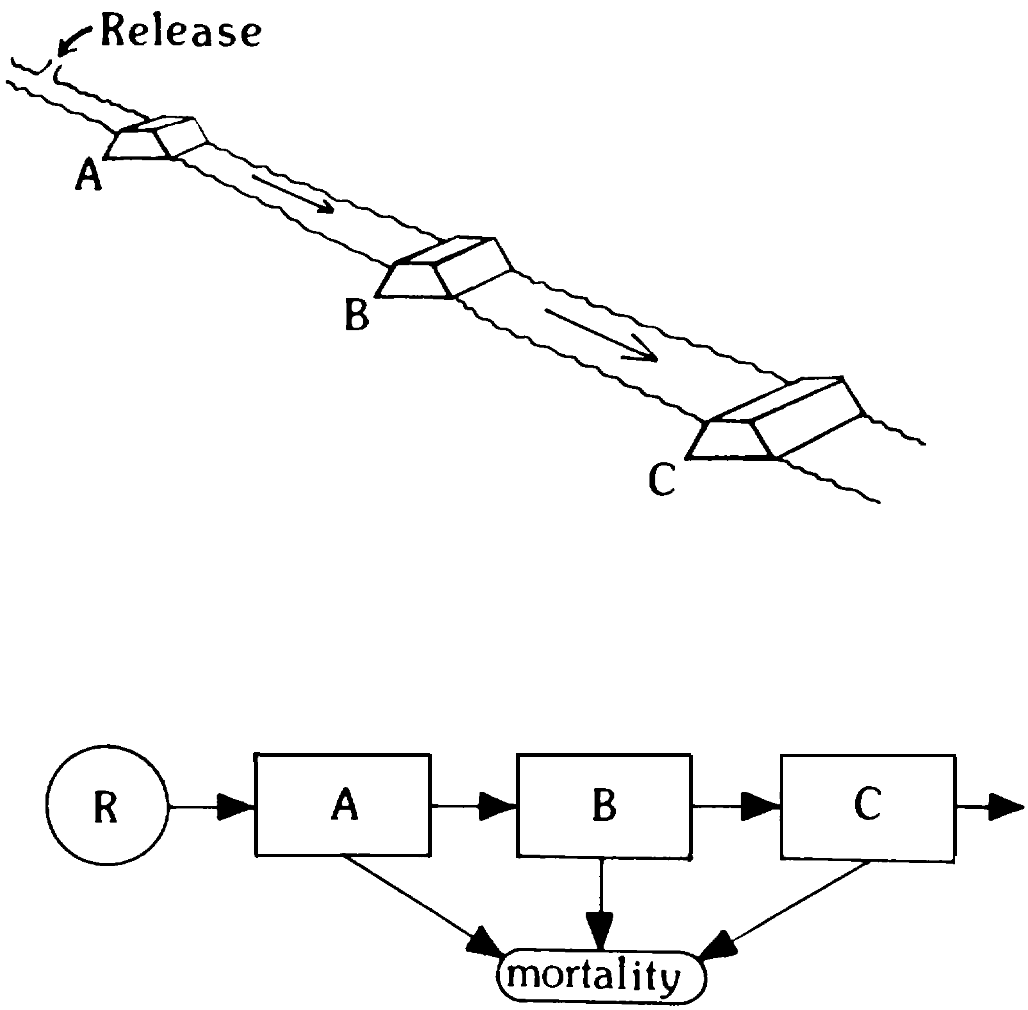


Figure 4.1 Three Reservoir System (above) Represented as a Conceptual Series of Compartments (below).

that the individual enters compartment $i+1$ at time t is

$$\begin{aligned}\text{Prob}(T_i = t) &= \text{Prob(survive } t \text{ time units in } i) \\ &\quad \times \text{Prob(traverse } i \text{ in } t \text{ time units)} \\ &\quad \times \text{Prob(survive boundary transition at time } t),\end{aligned}$$

or for convenience,

$$\text{Prob}(T_i = t) = f_i(t) = s_i(t) \times r_i(t) \times d_i(t). \qquad (4\text{-}1)$$

The conditional probability density function, $f_i(t)$, is referred to as the compartment function while the three functions to the right of the equality are called component functions. The first two component functions in Equation 4.1, $s_i(t)$ and $r_i(t)$, reflect natural mortality in the reservoir and rate of migration, respectively. The last component, $d_i(t)$, encompasses the mortality incurred in transition between compartments, for example, that associated with passing through a dam and the gauntlet of predators concentrated immediately downstream.

The compartment function can adequately describe the probability of a smolt leaving a reservoir on a particular day given that the fish entered the reservoir on some arbitrary day zero. In the general case one is more interested in following a group of individuals which may not enter the reservoir at the same time. Under these circumstances, the probability that a randomly selected smolt will leave the reservoir at a specified time is a convolution of the compartment function with a probability function associated with the reservoir entrance times. Consider a single compartment system composed of a single upstream input site, reservoir, and dam. Letting $p_0(v)$ be the probability density of a randomly selected individual entering the reservoir at time v, T be a random variable that measures the time of live passage past the dam, and time 0 defined as an arbitrary point in time prior to the first entrance to the reservoir, then

$$\text{Prob}(T = t) = \int_0^t p_0(v)\, f(t-v)\, dv. \qquad (4\text{-}2)$$

Since smolt migration is unidirectional (it's highly improbable that a smolt would migrate upstream past a dam), the sequential multi-compartment system can be conceptualized as a series of single compartment systems linked together. The output of an upstream compartment serves as the input for its adjacent downstream compartment. This suggests that the convolution (4-2) can be generalized to

$$\text{Prob}(T_i = t) = p_i(t) = \int_0^t p_{i-1}(v)\, f_i(t-v)\, dv, \qquad (4\text{-}3)$$

where $p_i(t)$ is the probability density for live passage from the ith compartment at time t. For the k compartment model, $p_0(\cdot)$ refers to the system input function, $p_1(\cdot), p_2(\cdot), \ldots, p_{k-1}(\cdot)$ are passage probability functions, and $p_k(\cdot)$ is the system output function.

The above formulation provides the framework that one can use to build a model of a sequential series of compartments. With minor modifications, the above relationships can also describe systems with multiple inputs and outputs for each compartment provided passage is irreversible. All that are required in either case are prescribed forms for the component functions and the system input functions. For simple systems, one can derive information about the system by using analytical approaches which permit analysis of the resultant passage probability functions and system output function. As the number of compartments and the complexity of the compartment functions increase, analytical solutions become effectively intractable. For these systems, numerical simulation using computers allows investigation of system behavior.

POSSIBLE FORMS FOR COMPONENT FUNCTIONS

Implicit or explicit in all existing models of smolt migration are prescribed forms for the component functions. As noted, most of the earlier efforts and discussion concerning this issue have focused on the dam mortality component, $d_i(\cdot)$; less attention has been paid to the problems of specifying reservoir mortality functions and travel time distributions. In the following discussion, potentially appropriate forms to define $s_i(\cdot)$ and $r_i(\cdot)$ are suggested. Though it is not necessary to require component functions in all compartments to share the same form, doing so can facilitate analysis. Since the present emphasis is on reservoir processes, complexities in dam mortality are ignored in the following discussion. In the derivation of an analytical solution, it is assumed that $d_i(\cdot)$ is simply a constant.

The component function, $s_i(\cdot)$, represents the probability of avoiding natural mortality. Natural mortality can be modeled as a Poisson process if two assumptions are made: (1) individuals are always at risk within a compartment, and (2) the magnitude of the risk within each compartment is constant throughout the migration period. Alternatively stated, all individuals are subject to a constant, instantaneous probability of dying during the entire time that they remain in each reservoir or river reach. If ρ = instantaneous mortality rate, and X = time spent in compartment preceding death, then

$$\mathrm{Prob}(X = t) = \rho\cdot\exp\{-\rho t\}. \tag{4-4}$$

Since $s_i(t)$ is equal to 1 - Prob{died in interval (O,t)),

$$\begin{aligned} s_i(t) &= 1 - \int_0^t \rho_i \exp\{-\rho_i x\} \; dx, \\ &= \exp\{-\rho_i t\}. \end{aligned} \tag{4-5}$$

Since there may be different levels of risk associated with each compartment or stock-specific differences in survivability, ρ and other parameters presented in this chapter are uniquely defined for each compartment and stock. Parameter subscripts are omitted to avoid cluttering the equations.

A different approach is used to select a distribution to describe travel time. The process of traveling through a reservoir can be conceptualized in a variety of ways. Each conceptualization may suggest a different probability density function based on theoretical considerations. The set of potentially appropriate distributions includes the exponential, lognormal, gamma, Weibull, and inverse Gaussian. Given this choice, it seems reasonable to choose the simplest form that is consistent with empirical data and provides flexibility in regard to both shape and scale. A relatively parsimonious distribution which meets these criteria is the three parameter gamma distribution. Using this form,

$$r_i(t) = [\Gamma(\alpha)]^{-1}\lambda^{\alpha}(t-\Theta)^{\alpha-1}\exp\{-\lambda(t-\Theta)\} \qquad t>\Theta, \tag{4-6}$$

where a, λ, and Θ uniquely define the shape, scale, and location, respectively, of the travel time distribution. In a biological context, λ and Θ reflect the speed with which the smolts migrate through each compartment. Hereinafter, λ is referred to as the travel rate (units are time^{-1}) and Θ is called the minimum travel time (in units of time). The dimensionless shape parameter, a, controls the skewness of the distribution which is a complex function of biotic and abiotic factors. As a approaches one, the travel time distribution approaches the exponential distribution; as a becomes very large, the distribution approaches the normal distribution.

ANALYTICAL SOLUTION

A thorough analytical analysis of a multiple compartment system is beyond the scope of this report. However, an analysis of a single compartment model provides an illustration of the methodology and utility of this approach. Based on this analysis, some implications for multiple compartment models are discussed. Consideration of the single reservoir model is made simple by assuming that all smolts enter the compartment at time 0 and dam survival equals unity (i.e., $d_i(t) = 1$, for all t). The focus of the analysis is on the time of passage through the reservoir and mortality within the reservoir.

Using equations (4-5) and (4-6) for $s(\cdot)$ and $r(\cdot)$,

$$p(t) = f(t) = \frac{\lambda^{\alpha}}{\Gamma(\alpha)} (t-\Theta)^{\alpha-1} \exp\{-[\lambda(t-\Theta)+\rho t]\} \qquad t>\Theta, \qquad (4\text{-}7)$$

which has the cumulative distribution function (CDF),

$$P(y) = \int_{\Theta}^{y} \frac{\lambda^{\alpha}}{\Gamma(\alpha)} (t-\Theta)^{\alpha-1} \exp\{-[\lambda(t-\Theta)+\rho t]\}\, dt \qquad t>\Theta. \qquad (4\text{-}8)$$

Rearranging,

$$P(y) = \frac{\lambda^{\alpha}\exp\{-\rho\Theta\}}{(\lambda+\rho)^{\alpha}} \int_{\Theta}^{y} \frac{(\lambda+\rho)^{\alpha}}{\Gamma(\alpha)} (t-\Theta)^{\alpha-1} \exp\{-(\lambda+\rho)(t-\Theta)\}\, dt \qquad t>\Theta. \qquad (4\text{-}9)$$

Equation (4-9) defines the probability of leaving the reservoir sometime during the interval Θ to y. Thus when examined in the limit, this CDF gives the fraction of the population which is expected to successfully migrate through the reservoir:

$$9 = \lim_{y \to \infty} P(y) = \frac{\lambda^{\alpha} \exp\{-\rho\Theta\}}{(\lambda+\rho)^{\alpha}}. \tag{4-10}$$

As demonstrated below, the compound parameter, ψ, is equivalent to the mean probability of passage across all individuals.

The mean probability of passage is used to determine the passage time distribution of those fish which survive passage through the reservoir. One impact of reservoir mortality within the model is that slower migrating fish are preferentially removed. This implies that the apparent rate of travel of fish surviving passage will be faster than the actual travel rate of the population as a whole. If p,(t) is defined as the probability of exiting the system at time t, given that an individual will successfully migrate, then

$$p_c(t) = p(t) \div \psi,$$

$$= \frac{(\lambda+\rho)^{\alpha}}{\Gamma(\alpha)} (t-\Theta)^{\alpha-1} \exp\{-(\lambda+\rho)(t-\Theta)\}. \tag{4-11}$$

Equation (4-11) is the probability density function for a gamma distributed random variable with scale parameter equal to $(\lambda+\rho)$, and shape and location parameters, α and Θ, respectively. The corresponding CDF for this distribution is denoted $P_c(t)$ and the mean and variance of the time of passage, T, are

$$\text{mean}(T) = \mu_T = \Theta + \frac{\alpha}{(\lambda+\rho)}, \tag{4-12}$$

and

$$\text{variance}(T) = \sigma^2_T = \frac{\alpha}{(\lambda+\rho)^2}. \tag{4-13}$$

Thus an increase in the mortality rate has the same effect on the transit time distribution of survivors as does an equivalent increase in the travel rate.

The distributions discussed thus far have the implicit assumption that the particles involved have a potentially infinite lifetime. Obviously such an assumption is inappropriate when biological organisms are involved.

In this case one may wish to truncate the migration period to the interval $(0,\tau]$, where τ is the longest observed or expected migration time. The probability of migrating sometime during the interval $(0,\tau]$ is equal to $P(\tau)$, while the conditional passage probabilities and the mean and variance of the time of passage differs than before. The truncated forms are:

$$p^*(t) = p(t) \div P(\tau), \qquad (4\text{-}14)$$

$$\text{mean}^*(T) = \frac{1}{P_c(\tau)}\left[\theta + \frac{\alpha}{(\lambda+\rho)} - \int_{\tau}^{\infty} t\, p_c(t)\ dt\right], \qquad (4\text{-}15)$$

and

$$\text{variance}^*(T) = \frac{1}{P_c(\tau)}\left[\frac{\alpha}{(\lambda+\rho)^2} - \int_{\tau}^{\infty} (t-\mu_T)^2\, p_c(t)\ dt\right]. \qquad (4\text{-}16)$$

As $P_c(\tau)$ approaches one, p*(t) approaches p,(t) and the mean and variance from each distribution become more similar. This may permit the simpler forms of μ_T and σ^2_T expressed in Equations 4.12 and 4.13 and the probability of passage derived in Equation 4.10 to be used as suitable approximations in most cases.

As previously noted, a sequential multi-compartment system can be viewed as a series of independent single compartment systems. This allows one to compute various indices of system behavior as functions of the individual compartment statistics. For example, the probability of passing through k sequential compartments is simply the joint product of the individual compartment passage probabilities;

$$\psi_{system} = \psi_1 \times \psi_2 \times \cdots \times \psi_k. \qquad (4\text{-}17)$$

Similarly, the mean and variance of time of passage through the system are linear functions of the corresponding statistics for each compartment;

$$\mu_{system} = \mu_1 + \mu_2 + \ldots + \mu_k, \qquad (4\text{-}18)$$

$$\sigma^2_{system} = \sigma^2_1 + \sigma^2_2 + \ldots + \sigma^2_k. \qquad (4\text{-}19)$$

When fish must pass through numerous reservoirs in sequence, such as in the Columbia River Basin, an analytical analysis of the compartment functions will produce a system output function defined by a multiple convolution of the basic compartment functions. As the distribution becomes more convoluted, it begins to approach the normal distribution. Thus, a reasonable approximation might be obtained as a product of the system passage probability, ψ_{system}, and the normal probability density function with mean = μ_{system}, and variance = σ^2_{system}.

Within the conceptual framework described above, not all individuals have the same probability of successfully migrating through a river reach because of the variation in time in transit. At times it may be useful to know the distribution of the probabilities of passage among individuals. For example, certain statistical models such as the release-recapture models for estimating treatment effects on survival described by Burnham et al. (1987) could be sensitive to the distribution of survival probabilities within experimental groups (more about this in Chapter 8). While the probability functions discussed thus far are all explicit functions of t, transformation techniques allow the calculation of useful probability density functions for variables other than time (see, for example, Mood et al. 1974:198-212).

To illustrate this technique, let Z = individual probability of passage, a continuous random variable with probability density function $h_z(\cdot)$, and let T = time in transit, a random variable with probability density function $h_t(\cdot)$. Also let Z be a function of time in transit such that Z = g(T), and T = $g^{-1}(Z)$. The theorem of transformations states that

$$h_z(Z) = h_t(g^{-1}(Z)) \, |d/dZ \; g^{-1}(Z)|. \qquad (4\text{-}20)$$

Thus if $g(\cdot)$ = equation (4-5), and $h_t(\cdot)$ = equation (4-6), then

$$h_z(Z) = \frac{1}{Z} \frac{(\lambda/\rho)^{\alpha}}{\Gamma(\alpha)} [-\log_e(Z) - \rho\theta]^{\alpha-1} \exp\{-(\lambda/\rho)[-\log_e(Z) - \rho\theta]\}, \qquad (4\text{-}21)$$

which has the domain (0,1) and is negatively skewed. The expected value of Z (E[Z]) is equal to ψ defined above (proof omitted). The variance of Z is equal to $E[Z^2] - \{E[Z]\}^2$, or

$$\mathrm{var}(Z) = \left[\left(\frac{\lambda}{\lambda+2\rho}\right)^{\alpha} \exp\{-2\rho\theta\} \right] - \psi^2. \qquad (4\text{-}22)$$

For most simulation studies, one can approximate equation (4-21) using a Beta distribution with the appropriate mean and variance.

PROJECTING IMPACTS OF MANAGEMENT ACTIONS

System managers are acutely interested in anticipating how mitigation measures and environmental fluctuations might affect reservoir passage. Events of both types influence passage via their direct impact on system parameters. The relative influence of each parameter on the passage probability function is reflected in the sensitivity of the mean probability of passage and of the mean and variance of time of passage to changes in parameter values. One measure of the sensitivity of each of these is their partial derivatives. Partial derivatives are defined as the instantaneous rate of change in a multi-parameter function with respect to a single parameter, assuming that all other parameters are held constant. Table 4.1 shows the partial derivatives of Equations 4.10, 4.12, and 4.13 with respect to each parameter and the sign of each derivative. The sign of the derivatives provide a quick but blurry view of how each parameter affects passage, and by implication the impacts expected from management actions. An estimate of the magnitude of the impact of a change in a parameter value can be assessed by examining the partial derivative more closely.

Management actions can be broadly divided into two types according to how they affect system parameters. Actions may be designed to speed the passage of smolts through the reservoirs by increasing the travel rate and decreasing the minimum travel time (type one), or designed to reduce the mortality rate (type two). Using the information in Table 4.1, the model suggests that type one actions such as the water budget increase the probability of passage while decreasing the mean and variance of travel time for the survivors. However, since the partial derivative of the probability of passage with respect to the travel rate is a nonlinear, decreasing function of travel rate, the magnitude of the expected increase in the probability of survival will decrease as the travel rate increases. A similar result in terms of improving the probability of passage is obtained for type two actions such as improving smolt quality or decreasing

Table 4.1 Partial Derivatives of the Mean Probability of Passage (ψ) and the Mean (μ) and Variance (σ^2) of Time of Passage with Respect to Each Parameter in the Model.

Parameter (p)	$\frac{\partial(\psi)}{\partial(p)}$	sign	$\frac{\partial(\mu)}{\partial(p)}$	sign	$\frac{\partial(\sigma^2)}{\partial(p)}$	sign
Minimum travel time (θ)	$(-\rho)\psi$	-	1	+	0	
Shape parm. (a)	$\log_e\left[\frac{\lambda}{\lambda+\rho}\right]\psi$		$\frac{1}{(\lambda+\rho)}$	+	$\frac{1}{(\lambda+\rho)^2}$	+
Travel rate (λ)	$\left[\frac{\alpha}{\lambda} - \frac{\alpha}{(\lambda+\rho)}\right]\psi$	+	$\frac{-\alpha}{(\lambda+\rho)^2}$	-	$\frac{-2\alpha}{(\lambda+\rho)^3}$	-
Mortality rate (ρ)	$-\left[\theta + \frac{\alpha}{(\lambda+\rho)}\right] \psi$	-	$\frac{-\alpha}{(\lambda+\rho)^2}$	-	$\frac{-2\alpha}{(\lambda+\rho)^3}$	-

predation on smolts. Decreasing the mortality rate also increases the probability of survival in a nonlinear fashion.

Since the partial derivatives of the mean and variance of the travel time with respect to travel rate and mortality rate are equivalent, one might expect changes in the travel time distribution of passage survivors which result from changes in travel rate to be obscured by fluctuations in mortality rate. However, this may not be a problem in real-world situations due to large differences in magnitude between travel rate and mortality rate. For example, if the travel rate is an order of magnitude greater than the mortality rate, then fluctuations in the mortality rate have a minimum effect on the mean and variance of travel time compared to the effect of changes in the travel rate. In contrast, fluctuations in the mortality rate can substantially affect the probability of survival even when the mortality rate is small relative to the travel rate.

System sensitivity to changes in parameter values are often best demonstrated graphically. If one establishes boundaries for the likely parameter space of a given reservoir or reach, then the effect of parameter changes on the system (as measured by dependent variables) can be explored using nomograms (Peterman 1975) and three-dimensional response surfaces. To construct a nomogram, all parameters except two are held constant. The remaining pair of parameters are used to define a plane in parameter space; contours or isopleths corresponding to distinct levels of system response are plotted within this plane. In a response surface, the dependent variable of interest is plotted relative to an axis which is perpendicular to the plane defined by the parameters. Response surfaces easily convey a sense of the nature of the system response to parameter changes but it may be difficult to extract precise values from looking at the graphs. When discrete levels of system response are of interest, nomograms may be more appropriate.

To illustrate these techniques, a nomogram for a single reservoir system is shown in Figure 4.2 and a response surface is shown in Figure 4.3 for the same system. In these examples, the minimum travel time and the shape parameter are fixed and the travel rate and mortality rate are permit ted to vary within certain bounds. The isopleths in Figure 4.2

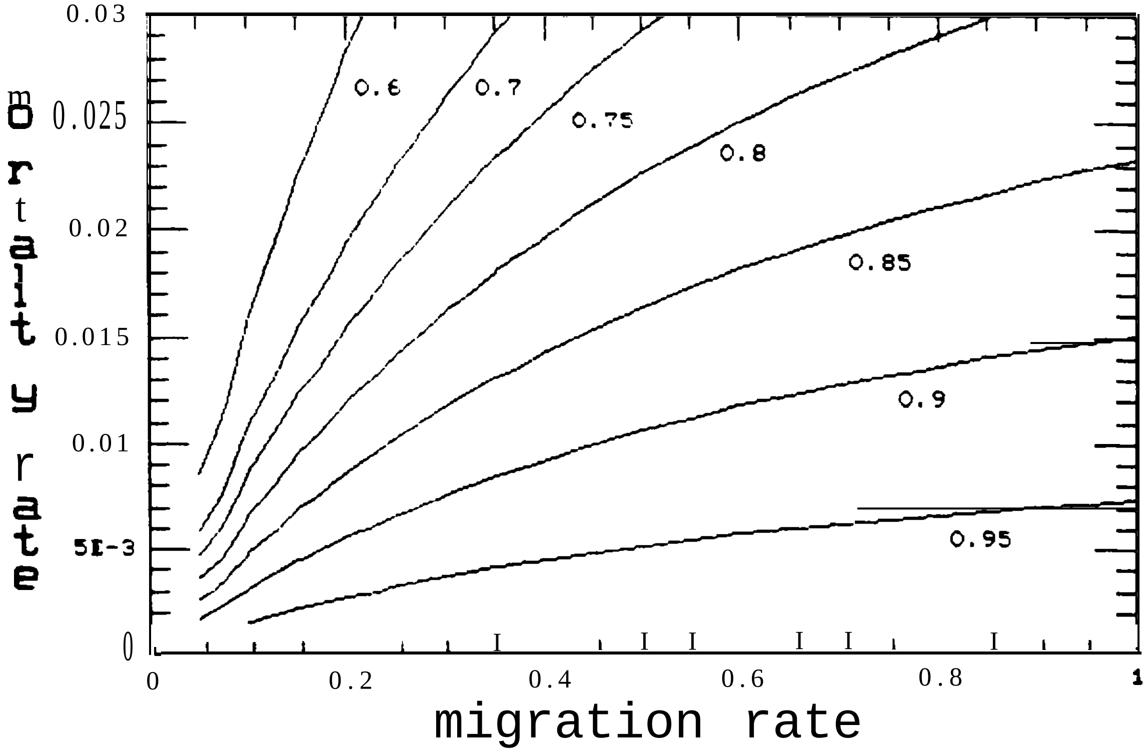


Figure 4.2 Nomogram for Single Compartment System. Isopleths correspond to discrete levels of the mean probability of passage.

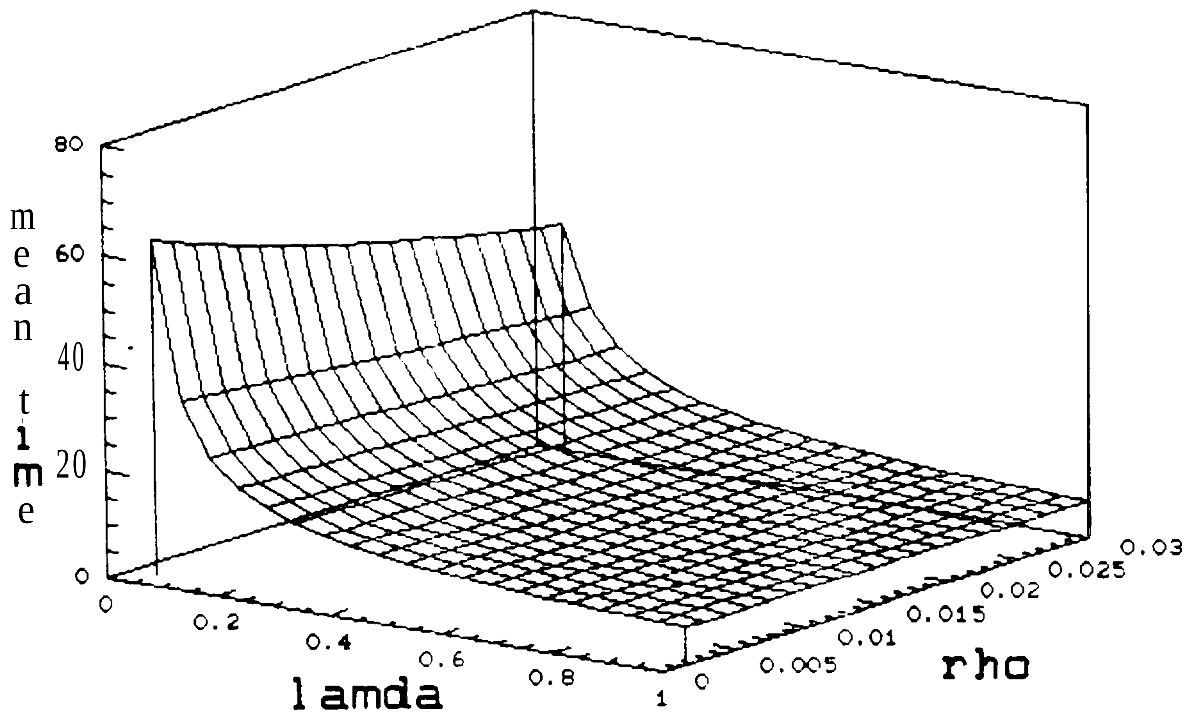


Figure 4.3 Response Surface for Single Compartment System Depicting Mean Passage Time as a Function of Migration Rate (Lamda) and Mortality Rate (Rho). .

correspond to distinct levels of the probability of passage which result from the various combinations of travel rate and mortality rate. Starting anywhere within the graph, a positive increase in travel rate leads to a higher probability of survival as one moves closer to a higher isopleth. The relative improvement in survival resulting from a increase in travel rate depends upon the value of mortality rate. The response surface depicted in Figure 4.3 illustrates the effect of travel rate and mortality rate on the mean time of passage. The curvilinear relationship between time of passage and travel rate is readily apparent. Mean passage time is relatively insensitive to the value of the mortality rate within much of the parameter space defined in the figure. Where the relative costs of altering system parameters are known, least cost approaches for moving from a given starting point to a more desirable isopleth can be determined using information inherent in nomograms and response surfaces and expansion path methods for production commonly applied in microeconomic analyses.

PARAMETER ESTIMATION

Regardless of the forms chosen for the component functions, accurate parameter estimation requires a suitable sample of compartment passage times and a reliable estimate of sampling efficiency. The ideal data set consists of exact measurements of time in transit for each sampled individual combined with accurate estimates of the probability of being sampled. Information from existing data sets, while less than ideal, may be adequate in some cases. Existing data sets usually contain daily counts of marked fish recovered at a dam or other sampling station. Counts of fish leaving a reservoir or reach during evenly spaced time intervals can be used in parameter estimation if the intervals are short in length relative to the entire migration period. For example, data collected at daily intervals is appropriate if the migration period covers several weeks but is insufficient if over 90% of the migration between sampling sites occurs in a few days. If the time intervals are excessively short and counts in each interval are extremely low, pooling data can reduce the information to a manageable number of observations. Erratic data can be smoothed either by pooling intervals or by calculating moving averages. Data which displays an unimodal distribution with non-negative skewness can generally be fit with the component functions discussed in Section 4.3.

Problems with current data arise because of imprecision and inaccuracy in the monitoring technologies which have been employed. The precise time required by each fish to travel from the release site to the sampling point is not always known since a large group of identically marked fish may be released over a period of several days. Also, estimates of daily sampling efficiency which are based on measurements of powerhouse discharge levels and duration of the sampling period have a high level of associated uncertainty. For example, roughly 60% of the variation in collection efficiency at McNary Dam for yearling chinook and steelhead could not be explained by variation in powerhouse discharge levels during the 1982 and 1983 outmigrations (estimate based on r values reported in Giorgi and Sims 1987). A further complication is that fish which are marked and released may not be physiologically ready to migrate and thus remain in the vicinity of the release site for several days or weeks before actively migrating. An initial step in the development of parameter estimates should involve a careful examination of existing data to identify a subset of the data which is likely to produce usable parameter estimates. Novel approaches for dealing with “messy” passage data should be further explored, including a review of techniques employed in disciplines other than fisheries where passage of particles through compartments is of interest. For example, Hughes and Matis (1984) discuss the application of a multicompartment model of ruminant digestion which is conceptually similar to the smolt passage model presented here and has similar parameter estimation problems.

An example application of a stochastic compartment model to passage data from a single reservoir illustrates some of the problems inherent in using existing data. Figure 4.4 demonstrates the fit of the component equations discussed above to a subset of mark-recovery data for yearling chinook salmon released immediately downstream of Priest Rapids Dam and recovered at McNary Dam (reported in McConnaha et al. 1985: Appendix I.B, p. 7). This data set, which is one of the better suited data sets for parameter estimation from the 1984 smolt monitoring program, represents only 2.2% of the total number of fish marked and released in this particular experiment and contains estimates of travel time for individual fish which may be wrong by as much as 5 days. Despite these factors, a

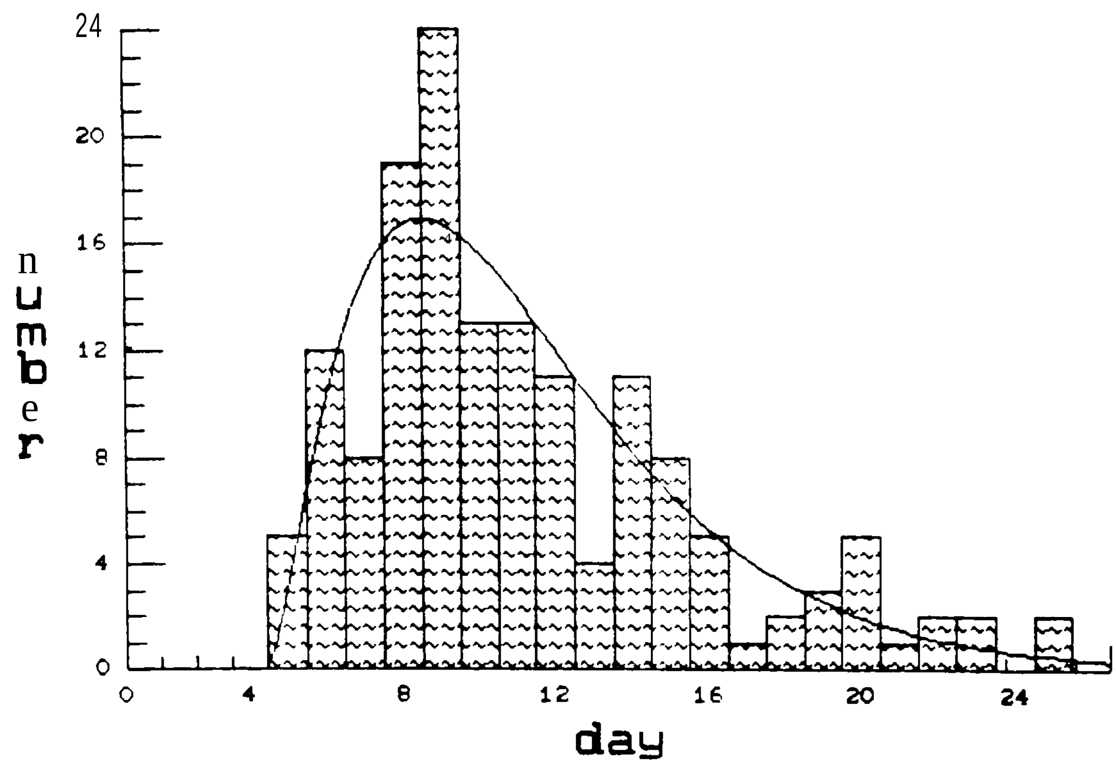


Figure 4.4 Component Equations Presented in Text Fitted to Mark-Recapture Data for Yearling Chinook Marked and Released at Priest Rapids Dam and Recaptured at McNary Dam. Number equals number of fish recaptured; day equals days between release and recapture. Data reported by McConnaha et al. 1985.

Kolmogorov-Smirnov goodness-of-fit test fails to reject the hypothesis that these data come from the hypothesized distribution at the 5 percent level of significance. Unfortunately, little can be confidently inferred from the estimated parameters because uncertainty in the estimate of sampling efficiency precludes a unique set of parameter estimates.

There is hope for better data in the future due to emerging technologies. A review of technological improvements in smolt monitoring is provided in Chapter 8. These advances in technology promise to be useful in a broad range of applications, including further model development. A combination of remote sensing tools such as passive integrated transponder (PIT) tags and hydroacoustics might provide the high quality data desirable for model implementation. Precise estimates of travel time should be feasible using PIT tags and continuous, remote sampling at dam bypass facilities (Prentice et al. 1985), while improved monitoring devices and statistical techniques might provide more reliable estimates of sampling efficiency. Continued development and assessment of these technologies is heartily encouraged.

Estimation of parameters for the component functions discussed above requires the use of non-linear methods. Unfortunately, non-linear parameter estimation as currently practiced is more of an art than an exact science. The proliferation of computers and statistical software provides a variety of methods; no single approach may claim to be consistently better. These methods generally employ algorithms which seek to iteratively minimize specified error functions by systematically altering parameter estimates. Users often have the option of selecting among error functions and defining the convergence criteria which determines when an acceptable solution has been reached. Initial parameter estimates must also be specified by the user. The final parameter estimates derived using these methods may be sensitive to the initial parameter estimates. This is particularly true when there is more than one local minimum in the error function, a situation that more often develops when several parameters are being estimated simultaneously. One should experiment with various combinations of initial estimates in searching for a global minimum. Nomograms can be used to formulate initial parameter estimates by delimiting the region in parameter space that could produce the observed sample statistics.

NUMERICAL SIMULATION

The process of developing a model which can be analyzed on a digital computer requires approximation of the continuous nature of the system using discrete analogs of the mathematical relationships. A first step is to divide the migration period into discrete time steps of equal size (e.g., hours, days, weeks). Having more steps leads to improved resolution at the cost of increased memory requirements and computational time. Compared to the uncertainty of natural systems, the error associated with lack of precision in any reasonable approximation is probably negligible. In the case of smolt migration in the Columbia River Basin, migration data are reported on a daily basis and day appears to be a satisfactory unit of time for simulation.

The following relationship is used when approximating the model:

$$\int_{(t-1)}^{t} \Phi(x)\,dx \simeq \tilde{\Phi}(t) \qquad t = 1,2,\ldots \,. \tag{4-23}$$

Where $\Phi(\cdot)$ is some arbitrary function continuous in the defined interval and $\tilde{\Phi}(\cdot)$ is its discrete analog. Using this notation, the passage probabi 1ity function (4-3) can be expressed in discrete form as:

$$\tilde{p}_i(t) = \sum_{y=0} \tilde{p}_{i-1}(y)\ \tilde{f}_i(t-y) \,. \tag{4-24}$$

This equation can be expressed using matrices:

$$\text{Let} \quad \tilde{p}_{i-1}(y) = a_Y \,,$$

$$\tilde{f}_i(t-y) = b_{t-y}$$

$$\text{and} \quad \tilde{p}_i(t) = c_t \,.$$

Then C = AB where A, B, and C are matrices defined as:

$$A_{(n+m+1)\times m} = \begin{bmatrix} a_0 & 0 & 0 & \cdots & 0 & 0 \\ a_1 & a_0 & & \cdots & 0 & 0 \\ a_2 & a_1 & a_0 & \cdots & 0 & 0 \\ . & : & : & \cdots & : & : \\ a_n & \cdot & \cdot & \cdots & 0 & 0 \\ 0 & a_n & \cdot & \cdots & 0 & 0 \\ 0 & 0 & a_n & \cdots & 0 & 0 \\ . & : & : & \cdots & : & . \\ 0 & 0 & 0 & --- & a_0 & 0 \\ 0 & 0 & 0 & --- & a_1 & a_0 \end{bmatrix},$$

$$B_{m\times l} = \begin{bmatrix} b_1 \\ b_2 \\ \\ b_m \end{bmatrix}, \qquad C_{(n+m+1)\times l} = \begin{bmatrix} c_1 \\ c_2 \\ \\ c_{n+m} \end{bmatrix}.$$

The dimension indexes n and m are arbitrarily chosen to reflect the last date of compartment input and longest reach passage time with probability substantively greater than zero. The obvious constraints on this model are that the elements of A and B must all be between zero and one and the sum of the columns within A and B must be less than or equal to one.

An advantage of numerical simulation is the flexibility to incorporate the added complexity and dimensionality which soon frustrates an analytical solution. The matrix formulation is easy to translate into computer code, allowing one to build a simulation model of a complex, multi-compartment system with little difficulty.

CONCLUSIONS

Three basic points developed in this chapter deserve emphasis:

- Reservoir passage and survival is a central concern in downstream migration which merits careful and intensive investigation. Mitigation measure to improve downstream passage through reservoirs are certain to be expensive and may be of unknown effectiveness.

- A stochastic compartment model approach provides a promising method of representing reservoir passage such that the effects of current mitigation measures can be evaluated and the impact of future actions might be anticipated. A notable strength of this approach is that it allows one to distinguish changes in rate of passage from changes in instantaneous mortality rate, two components that will be affected by mitigation measures in different ways.

- Existing monitoring data, combined with high-quality data which is expected to result from technological advances in smolt monitoring, can support implementation of the stochastic approach described herein.

Based on these premises, it seems prudent to pursue further development of a stochastic compartment model approach for modeling downstream passage. Development of this approach should proceed in conjunction with further improvement of downstream passage models such as FISHPASS such that the complementary strengths of alternative approaches can be exploited.

Chapter 5

MODELING THE ESTUARY AND EARLY OCEAN LIFE STANZA

INTRODUCTION

Each year, millions of juvenile anadromous salmon and trout pass through the Columbia River estuary on a journey to the Pacific Ocean where they grow to adulthood. For many of these salmonids, the relative success of the stock may be largely determined by the magnitude of the mortality incurred during their brief stay in the estuary or in the first few months following entry into seawater. Understanding of the mechanisms affecting survival and growth of juvenile salmonids in the estuary and near-shore oceanic environment is crucial to the effective management of these species.

Fishery managers and those interested in salmonid enhancement in the Columbia River Basin face the challenge of identifying management actions that will enhance the prospects of marine survival of each stock. While the list of available management options within the estuary and ocean is limited, managers can influence marine survival via upstream management actions which affect the timing of arrival, size, and physiological condition of smolts. Construction of applicable simulation models which might provide guidance is constrained, but not prohibited, by limited knowledge of the Columbia system. Despite a growing body of research on the Columbia and other Northwest estuaries, major uncertainties pervade our knowledge of critical ecological processes which affect salmonids during their estuary and early ocean (EEO) tenure. The term, “early ocean”, is used to refer to the period following migration from the estuary until the end of the year.

Where major uncertainties exist, a valid use of simulation models is to explore alternative hypotheses concerning causal mechanisms with the intention of identifying crucial data needs. In order to demonstrate the utility of developing models of the EEO component for Columbia Basin Stocks, an example model has been constructed which simulates the mortality and growth of a hypothetical chinook stock which enters the estuary as subyearlings. Mean length and mean time of arrival at the estuary can be adjusted to simulate the potential impacts of upstream management actions which affect these variables. The varied life history patterns among the anadromous species make it impractical to develop a single, generic simulation module for the EEO component of the salmonid life cycle that would be applicable to all species and stocks. Given the level of uncertainty surrounding our present knowledge of marine ecological processes, the intent of the model presented here is to focus debate and refine understanding.

BACKGROUND INFORMATION

Previous research which is especially pertinent to the Columbia River System includes work conducted by the National Marine Fisheries Service (Dawley et al. 1986; McCabe et al. 1983), information gathered under the auspices of the Columbia River Estuary Data Development Program (Lichatowich et al. 1984; Simenstad et al. 1984), and recent studies conducted off the coast of Washington and Oregon by Oregon State University personnel (Fisher and Pearcy 1988; Pearcy and Fisher 1988). The following information on the Columbia River Estuary, and the anadromous salmonids found therein, has been gleaned from these reports and through personal contacts with the authors, except where noted. Research on EEO concerns of other Northwest stocks is useful in terms of understanding ecological mechanisms. But one must be cautious in assessing the relevance of this research to the Columbia because of the distinct physiographical and bioenvironmental characteristics of the Columbia River estuary and adjacent ocean waters (see Pruter and Alverson 1972; Simenstad et al. 1984) and potential behavioral differences among salmonid stocks of different origin.

The Columbia River Estuary is conventionally defined as the lower 75 km of the river, ending with the jetties at the river mouth (Figure 5.1). It is a generally shallow area (<5 m in depth), with the exception of the major channels, and is laced with an extensive system of shallow channels, mud flats, shoals, and islands. Maximum seawater intrusion extends about 38 km into the estuary at low river flow but may be less than

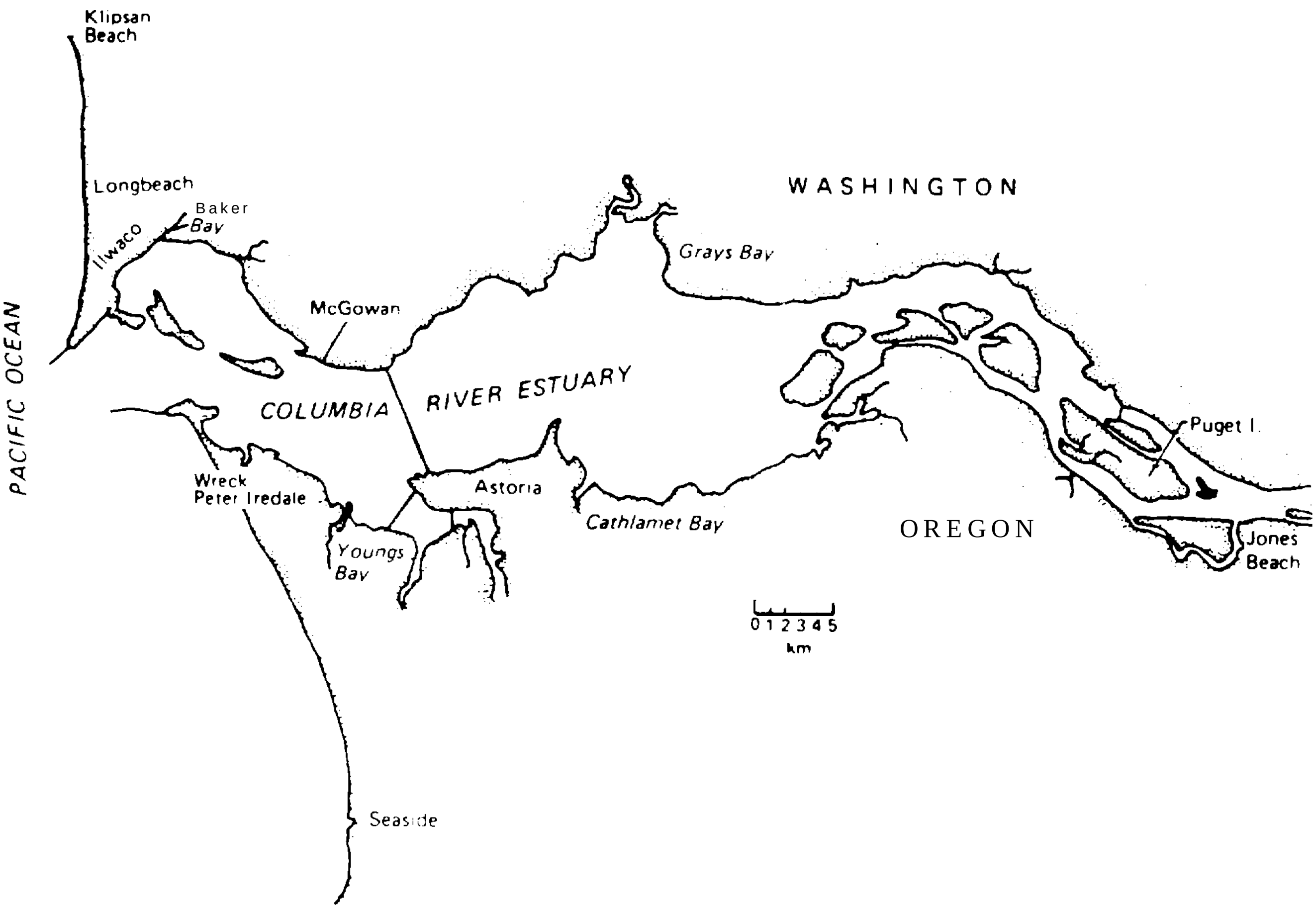


Figure 5.1 The Columbia River Estuary.

8 km during periods of high flow. There is an extensive and variable estuarine mixing area in the lower estuary depending on tide and river stage; the upper portion of the estuary is freshwater.

The character of the Columbia Estuary has changed dramatically in the past century. Extensive diking, filling, dredging and jetty construction have combined with the taming of the hydrologic regime which accompanied hydrosystem development within the Columbia Basin to alter the physical environment of the estuary, mainly through changes in the tidal prism and inriver sediment transport. Increased phytoplankton production within the lower Columbia reservoirs has probably lead to an increase in riverine input of organic carbon to the estuary and a decrease in inorganic nutrient input. It is not possible to identify changes in the estuary which are due solely to hydrosystem development separate from other factors, nor is it possible to separate the contribution of changes in the estuary to the historical decline in salmonid production from other causes.

Most Columbia River salmon and anadromous trout stocks apparently make little use of the estuary, moving quickly through the estuary and into the open ocean. The notable exceptions are fall chinook, summer chinook, and chum salmon which use the estuary as a rearing area, though the extent of this utilization is poorly understood. As a general rule, chinook salmon exhibit the most varied pattern of estuary use among the Pacific salmon (Healey 1982). Chinook salmon enter estuaries as either fry, subyearling fingerlings, or yearlings. Summer and fall chinook generally migrate to the Columbia Estuary as subyearlings after only a few months in freshwater. Subyearling chinook are the most numerous salmonid found in the estuary and spend more time there than do yearling migrants, which include spring chinook, coho, and sockeye salmon, and steelhead trout. Movement rates through the estuary are apparently both species and stock dependent, affected by where and when the fish begin their downstream migration.

Upon leaving the estuary, each species exhibits its own unique oceanic migration patterns with stock-specific variations. Chinook salmon and steelhead trout make extensive northward migrations during the first marine year (Hartt 1980) while many Columbia River coho salmon apparently remain in coastal waters off Washington and Oregon. These divergent migration patterns result in each stock being exposed to different environmental conditions affecting survival and growth.

OVERVIEW OF MODEL

The purpose of the model is to simulate the growth and survival of young chinook salmon from the time at which they enter the estuary through their first few months in the ocean. The model takes a fish stock distribution in terms of fish length and time of estuary arrival and maps this bivariate distribution into a univariate distribution of fish length at the end of the simulation period. Parameters which may be adjusted to reflect upstream management actions include size and timing of estuary arrivals and the estuary mortality rate (a partial reflection of smolt quality). Mechanisms are provided for changing the functional relationships among certain variables to incorporate alternative hypotheses.

The model structure is based upon two major assumptions that are inviolate in the model. First, residence time within the estuary is assumed to be based on fish length. There is evidence to suggest that larger fish migrate through the Columbia River Estuary at a faster rate (Dawley et al. 1986) and that resident subyearlings move out of estuaries as they grow larger (Simenstad and Wissmar 1984). The second major assumption is that growth in both the estuary and ocean varies with the seasons and is proportionate to body size. Fish growth is a complex function of numerous environmental variables which exhibit seasonal pat terns: river flow, water temperature, primary production, prey availability, et cetera. One approach to implicitly incorporate these factors into a model is to define a time dependent growth function which presumably adjusts for seasonal changes in the environment. The seasonal growth patterns in the estuary and the ocean may not coincide due to different environmental forcing functions.

MODEL CONCEPTUALIZATION AND STRUCTURE

The model is structured such that it tracks numbers and length distributions of fish in four compartments through time (Figure 5.2). Two

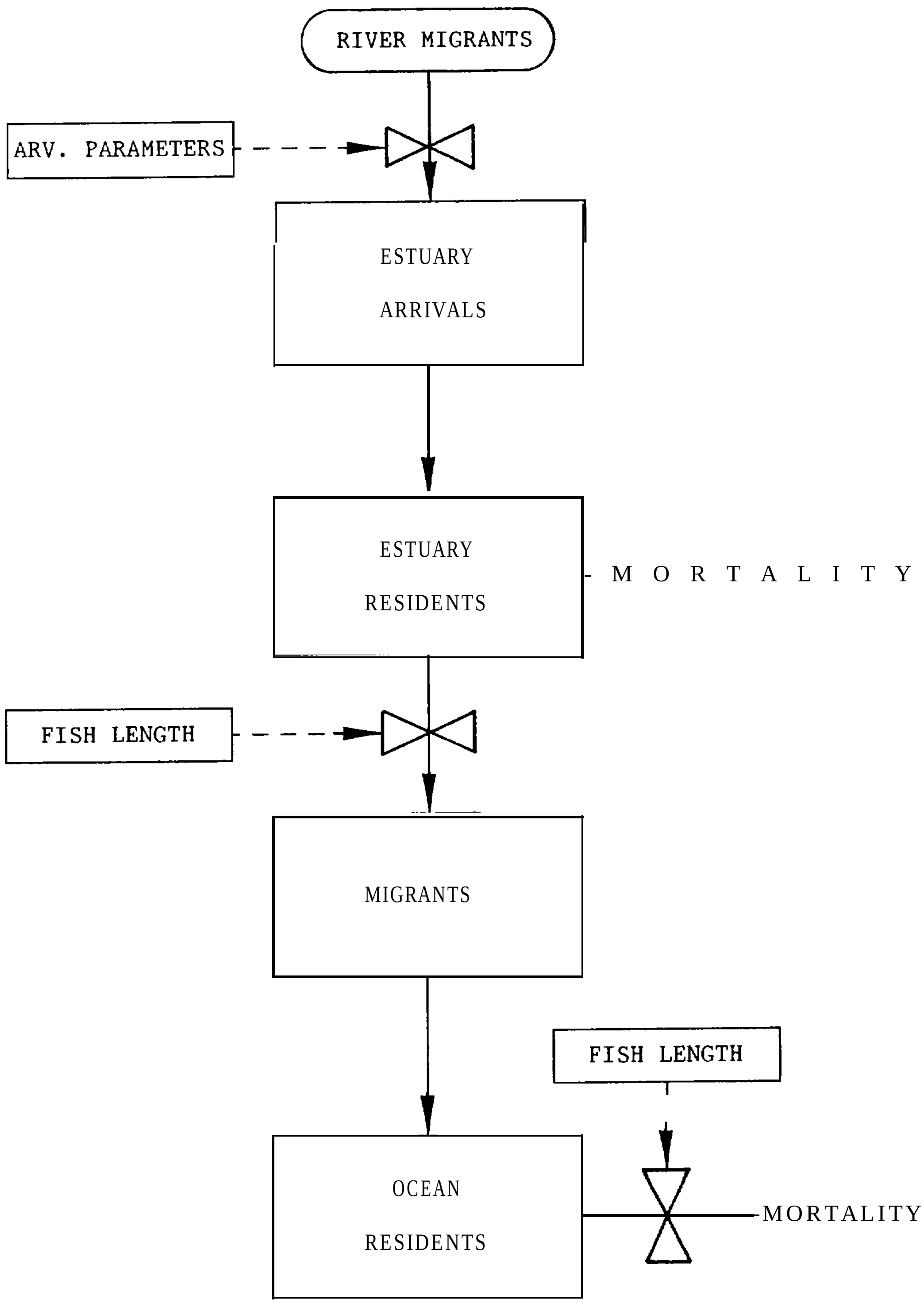


Figure 5.2 Conceptual Relationships Among Components of the Estuary and Early Ocean Model.

of these compartments, estuary arrivals and estuary migrants, are transient states where no changes occur in the individuals within them. The dynamic process of growth and mortality are simulated within the estuary and ocean residence compartments. Fish length is a key variable; it determines the proportion leaving the estuary and the survival rates of the ocean residents (as an option). The distribution of fish length (fork length) is assumed to follow a lognormal probability distribution as is appropriate when growth is proportionate to size (Boswell et al. 1979). Most of the model's internal calculations and many of the parameters are defined using the natural log transformation of fish length (transformed length).

Detailed descriptions of model components follow. The variable and parameter names used in the actual model (Table 5.1) are incorporated in the text where possible to alleviate confusion. Parameters, which refer to population attributes that do not change with time, and model constants are distinguished from variables by the use of upper case letters. Variable suffixes contained in brackets refer to time step. Where no brackets occur, the present time step, [t], is implicit. The number of days in each time step is defined with the constant, N_DAYS. Parameters and constants involving units of time must be specified in terms of days. The pound symbol (#) is used in the text as a prefix to denote intermediate variables used in calculations.

Estuary Arrivals

The input parameters given in Table 5.1 are used to characterize the number, timing, and length distribution of fish arriving at the estuary. The distribution of fish arriving over time is assumed to follow a normal distribution, ignoring the extreme tails. The range of arrival times is defined as the mean plus or minus 2.6 standard deviations:

$$\mathrm{ARV\_FD} = (\mathrm{ARV\_MT} - (2.6 \times \sqrt{\mathrm{ARV\_VL}})) / \mathrm{N\_DAYS}, \qquad (5\text{-}1)$$

$$\mathrm{ARV\_LD} = (\mathrm{ARV\_MT} - (2.6 \times \sqrt{\mathrm{ARV\_VL}})) / \mathrm{N\_DAYS}. \qquad (5\text{-}2)$$

The number of fish arriving at the estuary in each time step (arv_est), given t is within the interval, [ARV_FD, ARV_LD], is calculated as the product of the total population size (ARV_TOT) and the probability of

Table 5.1 Parameters and State Variables for Estuarine/Early Ocean Model

Input parameters (5)	
ARV_TOT	total number of fish reaching estuary
ARV_MT	mean time of arrival at estuary
ARV VT	variance of estuary arrival time
ARVL_BO	mean transformed length of arriving fish
ARVL_B1	Variance of transformed length of arriving fish
Control parameters (17)	
EST_SW	estuarine survival coefficient
EST_GBO	estuarine growth coefficient; mean of sine function
EST_GB1	" " " amplitude
EST GB2	" " " period
EST_GB3	" " " phase shift
ESTBL_BO	transformed reference length for growth model
ESTBL_B1	growth adjustment coefficient
CRT_ML	transformed critical migration length
MIGVL_BO	use depends on MIGVL_OPT (see text)
OCN_SVBO	ocean survival coefficient; intercept
OCN_SVB1	" " " slope
OCN_GBO	ocean growth coefficient; mean of sine function
OCN_GB1	" " " amplitude
OCN_GB2	" " " period
OCN_GB3	" " " phase shift
OCNBL_BO	transformed reference length for growth model
OCNBL_B1	growth adjustment coefficient
Model constants (4)	
ARV_FD	arrival date of first arriving fish
ARV_LD	arrival date of last arriving fish
PERIOD	length of simulation period
N_DAYS	number of days per time step

Table 5.1 Continued.

Independent state variable (1)

time	time step

Dependent state variables (23)

arv_est	number of fish arriving in estuary
arv_ml	mean transformed length of fish entering estuary
arv_mvl	variance of arv_ml
arv_vl	variance of transformed length of arriving fish
cum_arv	cumulative number of fish entering estuary
est_num	number of fish in estuary
est_g	estuary growth coefficient
est_ml	mean transformed length of fish in estuary
est_vl	variance of transformed length of estuarine fish
mig_est	number entering ocean
cum_mig	cumulative number of fish entering ocean
ocn_num	no. surviving first year that entered ocean at time t
ocn_svc	ocean survival coefficient
ocn_pct	percent surviving ocean that entered at time t
ocn_g	ocean growth coefficient
ocn_ml	mean transformed length of those entering ocean at time t
rel_sur	index of relative survival given ocean entry at time t
yrl_tot	total no. of ocean survivors at end of simulation period
yrl_sur	percent entering estuary that become ocean survivors
yrl_ml	population mean transformed length
yrl_vl	population variance of transformed length
sum_ml	summation variable used in calculating yrl_ml
sum_vl	" " " " " yrl_vl

arriving in the interval, (t-l, t], as calculated with the normal probability density function.

There are two options provided for calculating the mean (arv_ml) and variance (arv_vl) of transformed length of fish arriving at each time step. Using the default option, the mean and variance of arrivals are simply constant over time: arv_ml = ARVL_BO, and arv_vl = ARVL_B1, for all t. Optionally, one can assume that larger fish arrive at the estuary earlier than smaller fish. This might arise from larger fish migrating earlier in the season or from the larger fish in a hatchery release migrating faster than smaller members of the same cohort. The input parameters, ARVL_BO and ARVL_B1, are used under this option to denote the mean and variance of transformed length for the arriving population as a whole, aggregated over all time steps.

When the second option is chosen, arv_ml is estimated using the mean value predicted from a truncated normal probability density function. The limits of truncation depend upon the probability of arrival (P[t]) calculated in the determination of arv est. When the first migrants arrive at the estuary, the upper limit of truncation (ubound) is set:

$$\text{ubound}[\text{ARV\_FD}] = \text{ARVL\_BO} + (2.6 \times \sqrt{\text{ARVL\_Bl}}). \qquad (5\text{-}3)$$

The initial probability value (uprob) corresponding to ubound from the cumulative normal distribution function is equal to 0.9953. The lower limit of truncation (lbound) is calculated using the inverse of the cumulative normal distribution function and a probability value (lprob) equal to uprob - P[t]. With each successive time step, ubound is set such that ubound[t] = lbound[t-1], and lprob and lbound are calculated as above. The result is that the mean length of arriving fish (arv_len) progressively decreases over time.

The variance in length of arriving fish also varies with time under the second option. While the mean transformed length of arrivals is decreasing over time, the variance of this mean (arv_mvl) remains constant. A special subroutine estimates arv_mvl such that the aggregation of arriving fish across all time steps exhibits a variance in length equal to ARVL_B1. The variance in length of fish arriving within each time step is equal to the product of arv_mvl and arv_est. Thus the largest variation in length occurs among those groups of fish arriving during the center of the migration period and less variation among early and late arrivals.

Estuary Residents

The number of fish residing in the estuary during a given time period changes through migration and mortality. Mortality is represented within the model as a Poisson process such that the number of residents decreases exponentially in the absence of migration. The survival coefficient (EST-SUV) is defined as the natural log of the probability of surviving for a single day, each day being independent. (For example, if EST_SUV = -0.01, then the probability of surviving thirty days in the estuary would be equal to exp{-0.01 x 30) = 0.74.) The number of fish in the estuary (est_num) is adjusted in each time step for those entering the estuary (arv_est), leaving the estuary (mig_est), and dying:

$$\text{est\_num}[t] = (\text{arv\_est} + \text{est\_num}[t-1] - \text{mig\_est}) \times \exp\{\text{EST\_SUV} \times \text{N-DAYS}) . \qquad (5\text{-}4)$$

Estimation of mig_est and the length distribution of these migrants will be discussed in the next section.

The distribution of fish length within the estuary is similarly affected by migration and is also a function of growth during the time step. The mean transformed length (est_ml) is updated in a two step process. First, est_ml is adjusted for migration:

$$\#\text{est\_ml} = \frac{\left[(\text{est\_ml}[t-1] \times \text{est\_num}[t-1]) + (\text{arv\_ml} \times \text{arv\_est}) - (\text{mig\_ml} \times \text{mig\_est})\right]}{(\text{est\_num}[t-1] + \text{arv\_est} - \text{mig\_est})} . \qquad (5\text{-}5)$$

As larger fish migrate out of the estuary and smaller fish move in, the mean length of the fish in the estuary will decrease. This decrease in mean length due to migration will be offset to some extent by fish growth within the estuary. An adjustment for growth is made in the second step:

$$\text{est\_ml} = \#\text{est\_ml} + (\gamma \times \text{N\_DAYS}), \qquad (5\text{-}6)$$

where γ is the growth coefficient discussed below.

The growth expression used within the model is a modification of length (L) and weight (W) relationships commonly used in describing salmon growth (e.g., Ricker 1976). A commonly reported growth model for salmon assumes that the change in weight is proportional to current weight:

$$d\ W[t]\ /\ dt = W[t] \times g\ . \tag{5-7}$$

By substituting the length-weight relationship, $W = a \cdot L^b$, into equation (5-7), one obtains

$$d\ L[t]\ /\ dt = L[t] \times \gamma\ , \tag{5-8}$$

where $\gamma = g/b$. Inherent in this relationship are the assumptions that environmental conditions for growth remain constant and that the constant of proportionality is independent of fish size. In reality, the value of the coefficient, y, should vary over time to reflect changing environmental conditions and may also be correlated with fish length. For example, smaller fish might demonstrate a higher proportionate increase in length than larger fish under identical environmental conditions.

One way to incorporate such factors into the growth relationship is to express the growth coefficient as a function of fish length and environmental variables. If environmental conditions exhibit seasonal changes then time may be used as a surrogate variable for the environment. In the model, the coefficient from equation (5-8) is divided into two parts such that

$$\gamma = r \times (L_r\ /\ L[t])^{\alpha}\ , \tag{5-9}$$

where r is a variable growth rate, L_r is the fish length for which r is defined, and α is a parameter which reflects a change in growth rate as a function of the ratio, $L_r/L[t]$. If $\alpha = 1$, growth is linear; if $\alpha < 1$, the rate of change in length increases with increasing fish length; if $\alpha > 1$, growth rate declines as length increases, which is probably more common. The value of r represents the instantaneous growth rate of a fish of length L_r. The value selected for L_r should be close to the expected length of fish in the estuary, though any positive real number is permissible. Environmentally induced changes in growth rate are specified by altering the value of r while L_r and a remain constant. Within the model, values for r are calculated as a function of time using a harmonic function of the general form

$$r = \beta_0 + [\ \beta_1 \times \text{sine}\{(2\pi\ /\ \beta_2) \times (\text{time} - \beta_3)\}\ ], \tag{5-10}$$

where the parameters, β_0-β_3, are specific for a fish of length L_r. Since the model works with discrete time steps, the integrated form of equation (5-8) is used within the model.

Conceptually, the variance of length of estuary residents will also be affected by migration and growth. Moving fish of different lengths into and out of the population will clearly change the variation in length. Also, any random noise associated with γ will propagate changes in the variance structure. Tracking these changes can be a complicated process of dubious usefulness. The concept that is of more interest is the variance structure that determines the migration behavior or residence time of fish in the estuary--which may not be equivalent to the variance of estuary residents at a given time.

The approach taken within the model assumes that the variance measure of major interest is the "cumulative" variation in length of fish arriving at the estuary. Cumulative variance (est_vl) represents an aggregation of fish that are arriving at the estuary with those that have previously arrived. The calculation of est_vl is based on a working formula for variance (Steel and Torrie 1980):

$$\begin{aligned} est\_vl[t] = {} & \{\pi_a \times (arv\_ml^2 + arv\_vl)\} \\ & + \{\pi_e \times (est\_ml[t-1]^2 + est\_vl[t-1])\} \\ & - \{(\pi_a \times arv\_ml) + (\pi_e \times est\_ml[t-1])\}^2, \end{aligned} \tag{5-11}$$

where π_a = arv-est / (arv-est + est_num[t-1]), and $\pi_e = 1 - \pi_a$. Growth of fish in the estuary and outmigration of fish from the estuary affect est_vl indirectly through their impact on est_ml. An important determinant of the magnitude of est_vl is the difference between est_ml and arv_ml. The value of est_vl increases at a faster rate with larger- differences between est_ml and arv_ml.

Estuary Migrants

The model incorporates the assumption that fish begin to leave the estuary as they attain a certain minimum length, driven by an inherent urge to migrate. Previous studies have suggested that larger- fish move quickly out of the estuary and into the open ocean, and then begin a lengthy migration to northern rearing areas. The motivation for leaving the estuary may be to search for larger, more desirable prey items, or as an adaptive mechanism to reduce intraspecific competition. Physical factors such as water temperature may also have a role in prompting migration.

In the model, the number leaving the estuary during each time step is calculated as the product of the number of fish in the estuary and the probability of being longer than the specified minimum length. This probability of migrating is estimated using the cumulative normal distribution function with mean, est_ml, and variance, est_vl, and the transformed critical migration length, CRT_ML. The mean transformed length of the migrants (mig_ml) is estimated using this same normal distribution truncated at CRT_ML and est_ml plus 2.6 standard deviations.

There are three options for defining the variance of transformed length of the estuary migrants (mig-vl). In the first option, mig_vl is a constant defined by the user. In the second option, mig_vl is equal to the product of est_vl and a user-defined constant. The third option assigns a value for mig_vl that is proportionate to est_vl and the probability of migration. Under this option,

mig_vl = est_vl[t-1] x (mig_est/est_num[t-1]) x MIGVL_BO, (5-12)

where MIGVL_BO is a user-defined constant. The concept of migrant variance is important because of its impact on the final length distribution of the population at the end of the simulation period. Unfortunately, there seems to be no empirical evidence or theoretical justification for preferring one option over the others. It is entirely possible that the variance of length of estuary migrants in the real system is a random variable with little resemblance to either of the values offered in the model.

Ocean Residents

Growth and survival of fish in the ocean is simulated differently than for the estuary residents. Whereas with the estuary residents the model aggregates individuals entering the estuary with fish present in the estuary at the beginning of each time step, subpopulations of fish entering the ocean at different times remain segregated until the end of the simulation period. The reason for this is that the model incorporates size-dependent growth and mortality in the ocean and it is more convenient to model these processes using subpopulations that are more homogeneous relative to body length than the parent population.

Each subpopulation enters the ocean with an initial mean length and variance determined in the estuary migrant compartment (discussed above). The mean length increases over time as a function of growth. A growth model, equivalent to that used for estuary residents, is defined using ocean-specific parameters. The variance of transformed length is assumed to remain constant over time: ocn_vl = mig_vl. The variance in fork length increases as the mean length increases due to the relationship between the fork length and transformed length distributions. Recall that if a random variable, X, is distributed normally with mean = μ, and variance = σ^2, and Y = exp{X}, then Y follows a lognormal distribution and E(Y) = $\exp\{\mu+(\sigma^2/2)\}$, and $Var(Y) = \exp\{2\mu+\sigma^2\}(\exp\{\sigma^2\}-1)$.

The model allows one to define ocean mortality such that the percentage of each subpopulation surviving through each time step increases (or decreases) as the mean length increases. The survival coefficient for the exponential decay model of survival, similar to that used in the estuary, is defined for the ocean residents as a function of fish length:

$$ocn\_svc = OCN\_SVB0 + (OCN\_SVB1 \times \exp\{ocn\_ml\}). \quad (5\text{-}13)$$

The proportion surviving each time step is equal to exp(ocn_svc}, where ocn_svc must be less than or equal to zero. The parameter, OCN_SVB1, determines the relationship between length and survival. If OCN_SVB1 is positive, survival increases with length. If OCN_SVB1 = 0, then survival is independent of length.

Because of the structure of the model algorithm, the variables having the prefix, ocn , which are output by the model at each time step, reflect terminal statistics specific to the subpopulation that entered the ocean during that time step. The relative index of survival (rel_sur) is a comparative measure of the expected mar-ine survival of the subpopulation entering the ocean at time t, assuming all fish entered the estuary at time ARV_FD. Variables with the prefix, yrl_, refer to the entire population surviving the simulation period and as such are meaningful only at the conclusion of the final time step.

EXAMPLE APPLICATION

For purposes of illustration, the model was applied to a fictitious chinook salmon stock. Parameters values used for the benchmark simulation (Table 5.2) characterize a population which has a mean arrival date at the estuary of April 1, and a mean arrival length of 57 mm. A simulation period of one year, January 1 to December 31, was used with a time step of 5 days. Where options were available within the model, the more complex options were chosen for the benchmark simulation. Three assumptions were stipulated by the use of these options: (1) fish arrive at the estuary according to length, (2) the variance in length of the estuary migrants is proportional to the percentage migrating, and (3) mortality in the ocean is length-dependent. It was assumed that environmental fluctuations in the estuarine and oceanic growth coefficients follow different patterns (Figure 5.3).

A crucial assumption reflected in the parameter values defined for the benchmark and subsequent simulations is the position that the estuary is a refuge from predators for small salmon. This view of the Columbia River Estuary is proposed by McCabe et al. (1983) based on their analysis of the inter-relationships between juvenile salmonids and non-salmonid fish. The potential impact of piscivorous birds or mammals is less understood. The concept of the estuary as a refuge is incorporated in the model by defining survival rates for estuary residents to be higher than survival rates for ocean residents of similar size.

Results from the benchmark simulation are presented as an example of model outputs and should not be interpreted as being indicative of any real stock. Fish began arriving at the estuary in late January and continued to arrive until mid June (Figure 5.4). The distribution of estuary migrants over time was more skewed than the distribution of estuary arrivals, and the peak in estuary migrants lagged the peak in estuary arrivals by approximately 70 days. Fish began leaving the estuary around May 1 and migration continued through the end of the year. Of the simulated fish population that arrived at the estuary, by the end of the year 46.1% had died in the estuary, 41.4% had died in the ocean, 2.8% remained alive in the estuary, and 9.7% remained alive in the ocean.

Table 5.2 Parameter Values and Constants Used in Benchmark Simulation.

Parameter	Value	Units*
ARV_TOT	100,000	count
ARV_MT	90	days
ARV_VT	853	$(days)^2$
ARV_FD	14	days
ARV_LD	166	days
ARVL_BO	4.0	$\log_e(mm)$
ARVL_B1	0.08	$[\log_e(mm)]^2$
EST_SUV	-0.0052	$(days)^{-1}$
EST_GBO	0.005	$(days)^{-1}$
EST_GB1	0.003	$(days)^{-1}$
EST_GB2	548	days
EST_GB3	0	days
ESTBL_BO	4.0	$\log_e(mm)$
ESTBL_B1	0.5	NA
CRT_ML	4.78	$\log_e(mm)$
MIGVL_BO	0.2	NA
OCN_SVBO	-0.0202	NA
OCN_SVB1	0.000044	$(mm)^{-1}$
OCN_GBO	0.004	$(days)^{-1}$
OCN_GB1	0.003	$(days)^{-1}$
OCN_GB2	657	days
OCN_GB3	36.5	days
OCNBL_BO	2	$\log_e(mm)$
OCNBL_B1	2	NA
PERIOD	365	days
N_DAYS	5	days

*NA = not applicable; dimensionless par-ameter.

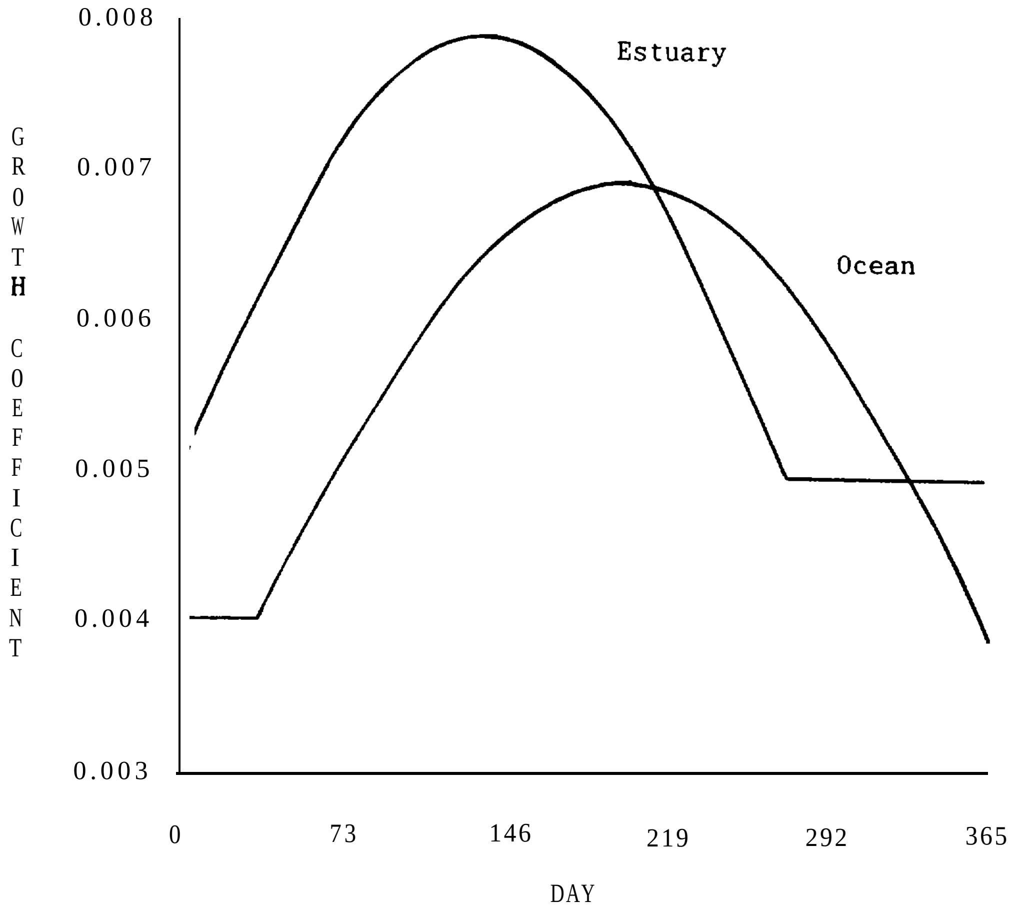


Figure 5.3 Time Series of Growth Coefficients for Estuary and Ocean Residents Used in the Model. The references lengths for the estuary and ocean growth coefficients were 55 and 181 mm, respectively.

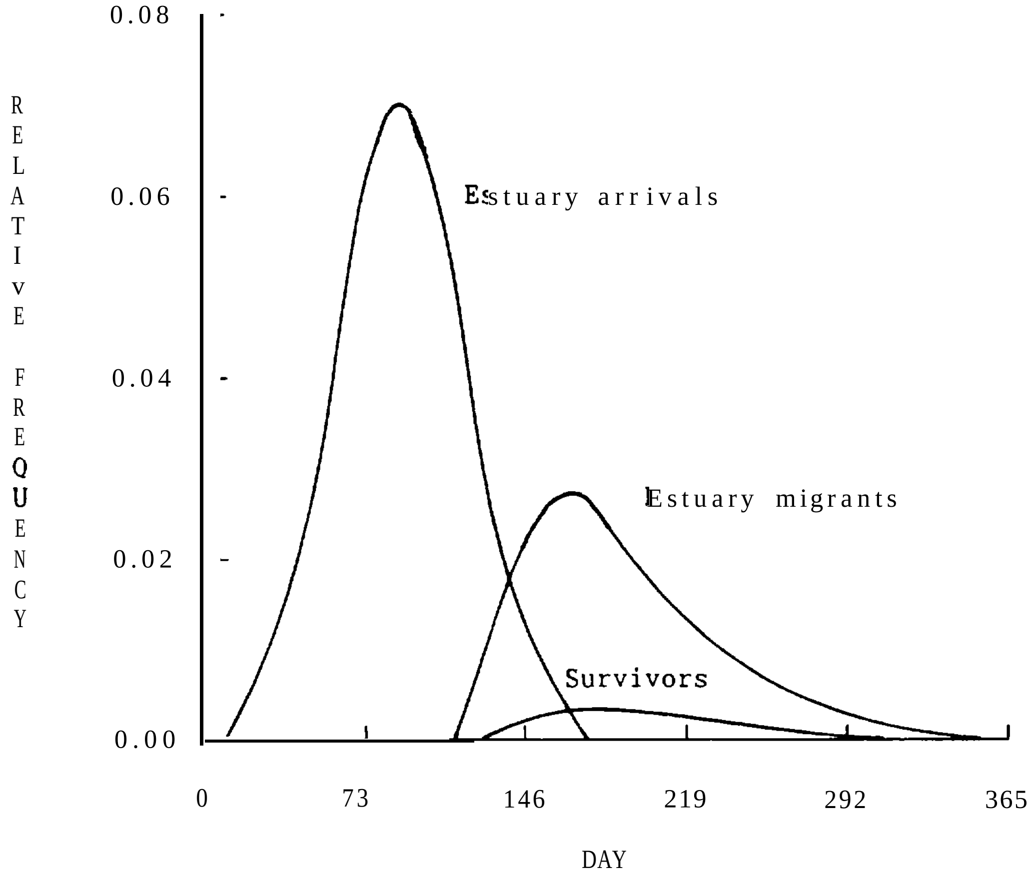


Figure 5.4 Relative Frequency Distribution of Estuary Arrivals and Estuary Migrants Over Time, and Ocean Survivors Plotted Versus Date of Ocean Entry. The vertical axis is standardized such that the area under the curve denoted "estuary arrivals" is equal to one.

Using the option which calls for larger fish to arrive earlier, the mean length of arrivals decreased in an almost linear fashion (Figure 5.5). The mean length of estuary residents increased due to growth in the first part of the year but then decreased slightly as the larger fish migrated out of the estuary and estuary growth rates slackened. The trend in mean lengths of the estuary migrants essentially paralleled that of the estuary residents, with a difference of about 50 mm. At the end of the simulation period, the ocean residents exhibited a mean length of 273 mm with a standard deviation of 64.5 mm. This represents an increase in mean length of 216 mm and a four-fold increase in the standard deviation of length since the time of estuary arrival (Figure 5.6).

In order to demonstrate the potential impacts of upstream management actions on EEO survival and growth, the mean length and mean time of arrival were varied systematically. Five levels of mean length of arrivals (ARV_MLEN = 26, 41.5, 57, 72.5, and 88 mm) and five levels of mean date of arrival (ARV_MT = 60, 75, 90, 105, and 120 days) were used in combination to construct a 5x5 design matrix. Mean length and time of arrival are two parameters that, at least for hatchery stocks, are moderately pliant. All other parameters from the benchmark simulation were held constant to simplify the analysis. Note that keeping the estuary survival rate constant assumes that smolt quality is independent of mean fish length and time of estuary arrival. This assumption is probably unrealistic for the real system (Wedemeyer et al. 1980; Mahnken et al. 1982). The length of the simulation period (PERIOD) for each model run was set equal to ARV_MT plus 275 days to insure that each population spent the same amount of time in the system. Otherwise, direct comparisons of final population sizes would be misleading.

Simulation results from using each combination of parameters were used to describe response surfaces for selected output variables. The response surfaces generated for summary variables at the end of the simulation period provide some conceptual information that might help explain or anticipate the relative success of hatchery release practices. The response surface generated for the adjusted number of ocean residents at the end of the year suggests considerable interaction between arrival size

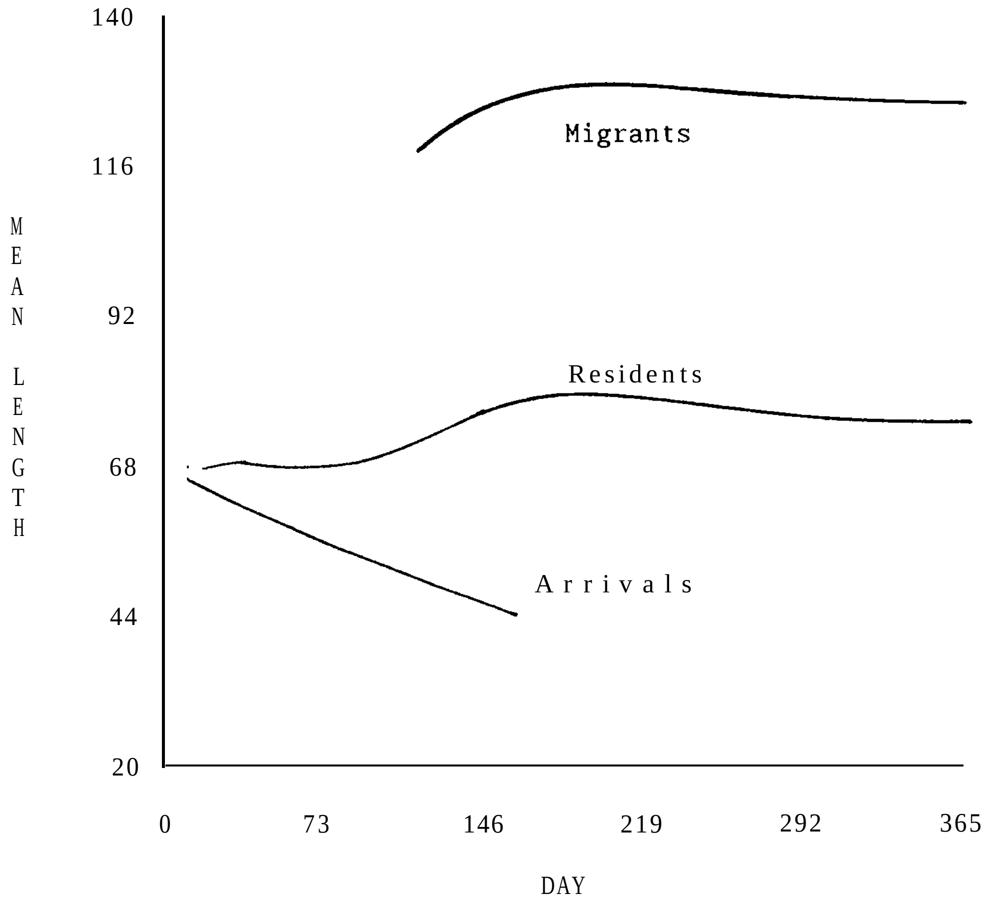


Figure 5.5 Mean Lengths (in mm) of Fish in the Estuary (Residents) and of those Fish Entering (Arrivals) and Leaving (Migrants) the Estuary Over Time.

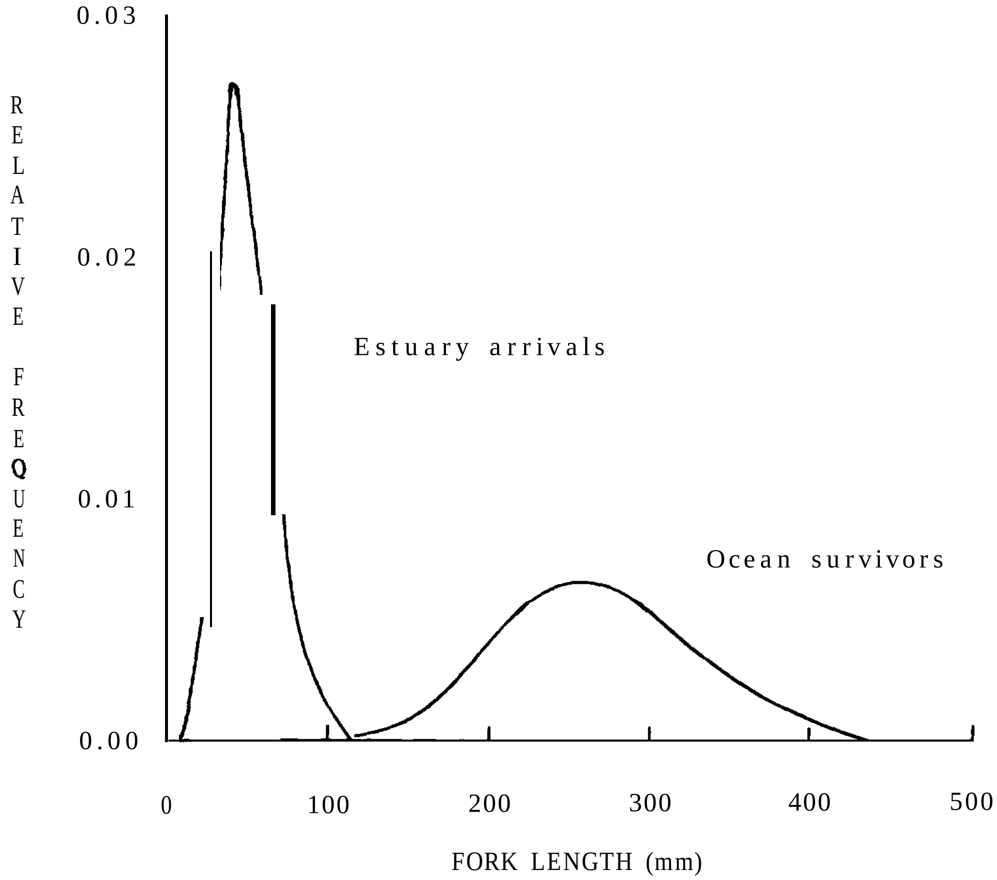


Figure 5.6 Relative Frequency Distributions of Fish Length at Time of Estuary Arrival and of those Fish Residing in the Ocean at the End of the Simulation Period. The total area under each curve is equal to one.

and date of arrival (Figure 5.7). Medium-sized fish appear to have an comparative advantage during the earliest arrival dates but this advantage shifts to the largest fish as the mean date of arrival increases. This suggests that it may be disadvantageous to either leave the relative protection of the estuary too soon or stay in the estuary too long. The earlier arrival dates produce more ocean survivors because the fish are able to take advantage of better conditions for growth in the estuary and ocean.

Even more dramatic than the impact of parameter changes on ocean survivors was the response in the number of fish remaining in the estuary at the end of the simulation period (Figure 5.8). The simulated variation in the number of estuary "residuals" was an order of magnitude greater than the variation in ocean survivors. The extent of the variation which can be ascribed to different mean lengths at arrival swamps the more limited variation due to different arrival times. The number of estuary residuals appears to drop exponentially as the mean length of arrivals increases while later arrival dates lead to a more modest linear increase in residuals. Size at arrival is apparently the more important determinant of estuary residence time within the model.

The length distribution of ocean residents at the end of the simulation period is also affected by arrival length and date of arrival. The response of mean length of ocean residents to the parameter changes is described by the plane:

$$\text{mean length} = 219.5 + 1.3(\text{ARV\_MLEN}) - 0.27(\text{ARV\_MT}), \qquad (5\text{-}14)$$

which was fit using multiple regression ($R^2 = 0.996$). As one might expect intuitively, terminal mean length increases with increased length at arrival and decreases as mean date of arrival increases. The response surface for the standard deviation of length of ocean residents is more complex than that for length, is not easily explained, and provides a good example of the nonlinear interactions occurring among variables within the model (Figure 5.9).

A limited attempt was made to look at the relationships between assumptions made in the model and conclusions which might be drawn from the

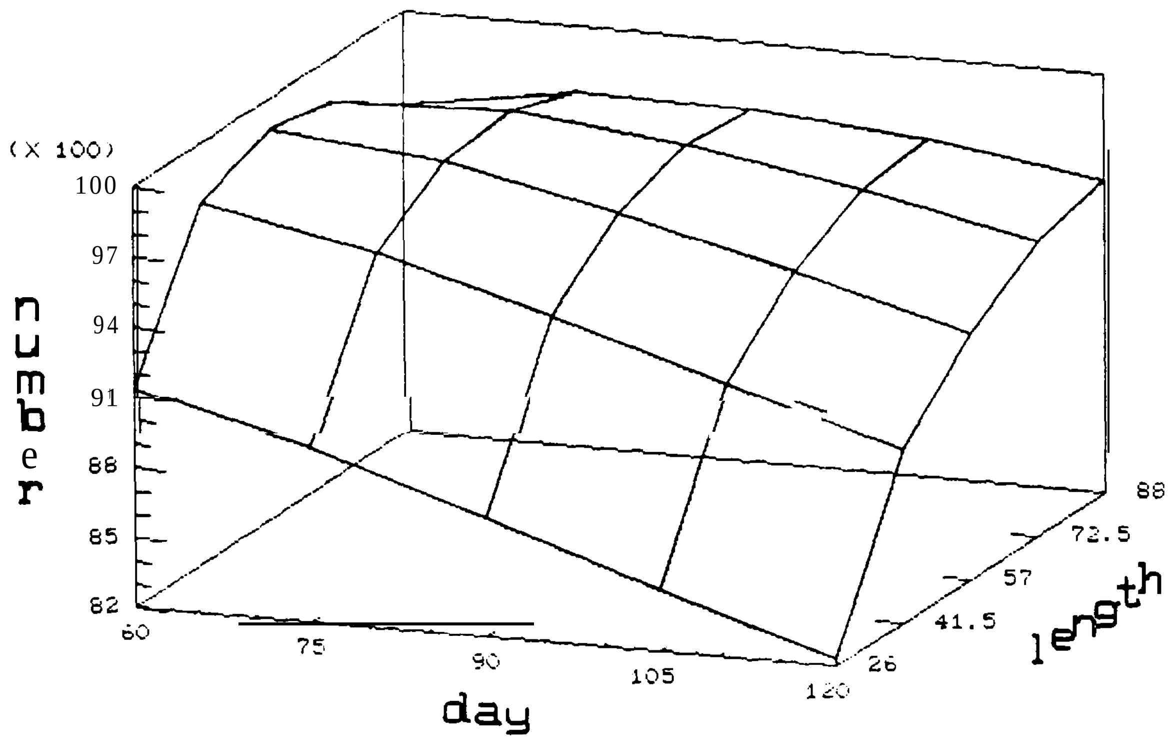


Figure 5.7 Response in Total Number of Ocean Residents (Number) at the End of the Simulation Period to Changes in Mean Arrival Date (Day) and Mean Length (Length) on Arrival at the Estuary, Assuming that Mortality in the Ocean is Size-Dependent.

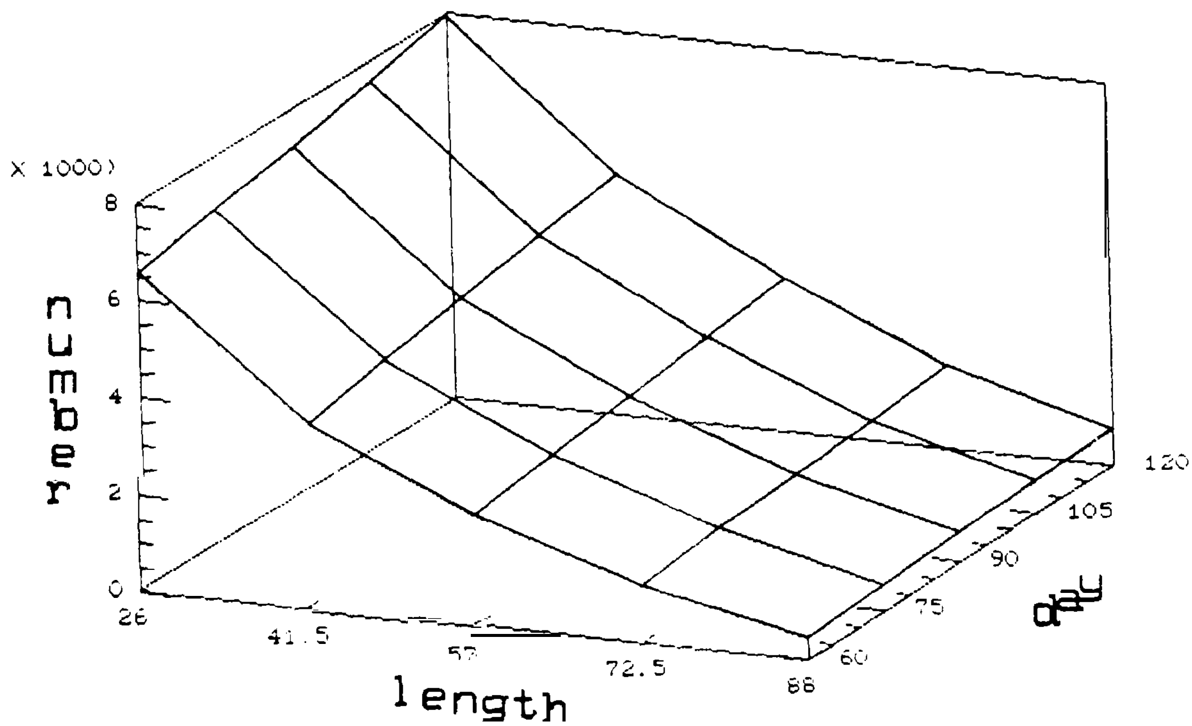


Figure 5.8 Response in Total Number of Fish Remaining in the Estuary at the End of the Simulation Period to Changes in Mean Length (Length) and Mean Date (Day) of Arrival at the Estuary.

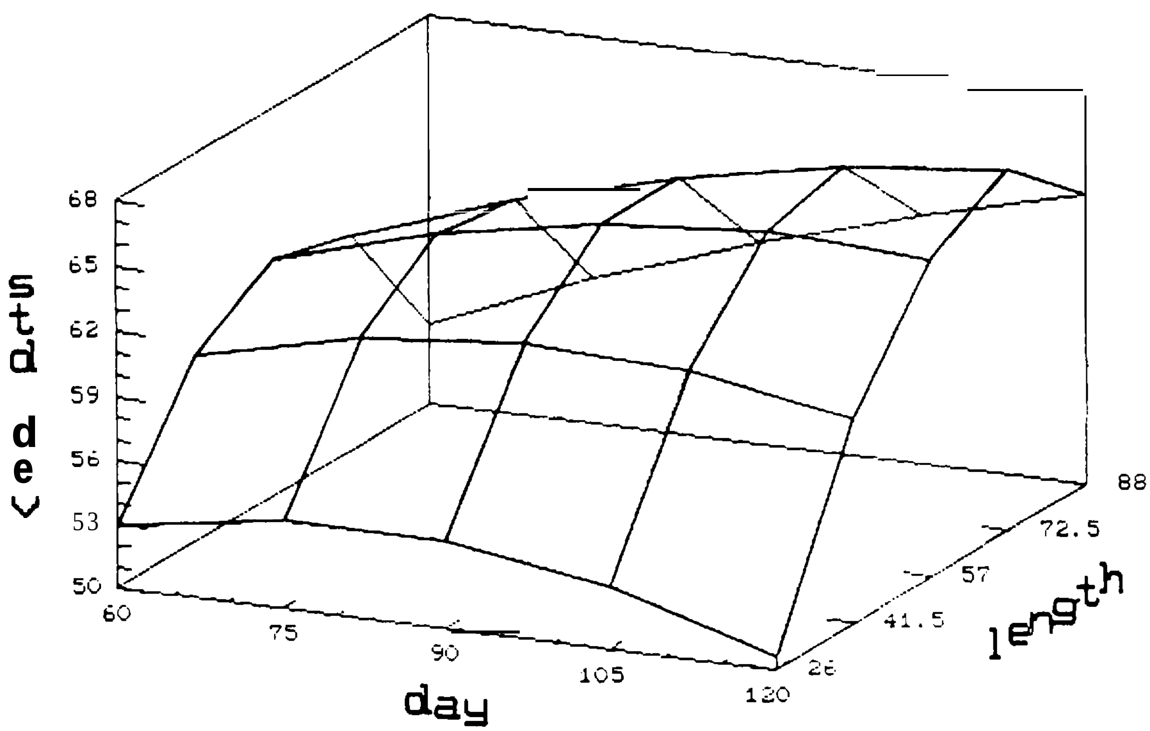


Figure 5.9 Response in the Standard Deviation of Length of Ocean Residents (Std Dev) at the End of the Simulation Period to Changes in Mean Arrival Date (Day) and Mean Length (Length) on Arrival at the Estuary.

model's application. A noteworthy result was that the response surface for adjusted number of ocean survivors shown in Figure 5.7 was found to be highly dependent on the assumption that mortality in the ocean is size-dependent. Holding ocean mortality constant for all fish (ocn_svc = -0.01) resulted in a dramatic change in the response surface (Figure 5.10). The comparative advantage under this scenario is now given to the smaller fish. This results because smaller fish spend a higher proportion of time in the estuary where the survival rate is now sufficiently higher than that in the ocean to create an apparent advantage. One could reverse the situation such that mortality rates are assumed to be higher in the estuary than in the ocean. Under these conditions, fish which enter the estuary at a larger size would have an advantage in survival.

DISCUSSION

In the effort to develop plans for life-stanza models, the estuary and early ocean component has been one of the more difficult to resolve. It is always difficult to step into an unfamiliar arena and try to model a system about which one knows very little. The usual approach is to begin by evaluating models which may be applicable to the system at hand with the hope of building on the strengths and weaknesses of the prior efforts. When such example models are lacking, the best alternative is to start with a clear conceptual understanding of the major processes at work within the system. When even the fundamental task of defining a conceptual model of the system is problematic, one naturally questions the feasibility of building a useful simulation model. Is it worth the effort?

In spite of considerable uncertainty concerning ecosystem processes, progress made in developing the heuristic model presented here suggests that building species-specific models of the EEO component of the Columbia River System is feasible, and can be instructive. The present module can be viewed as an essential first step in the development process. The model in its present condition is not an appropriate end product. Much evaluation and validation remains to be done. Let the present model serve only as an example of a realistic modeling approach and of the potential value of having valid models of the EEO component.

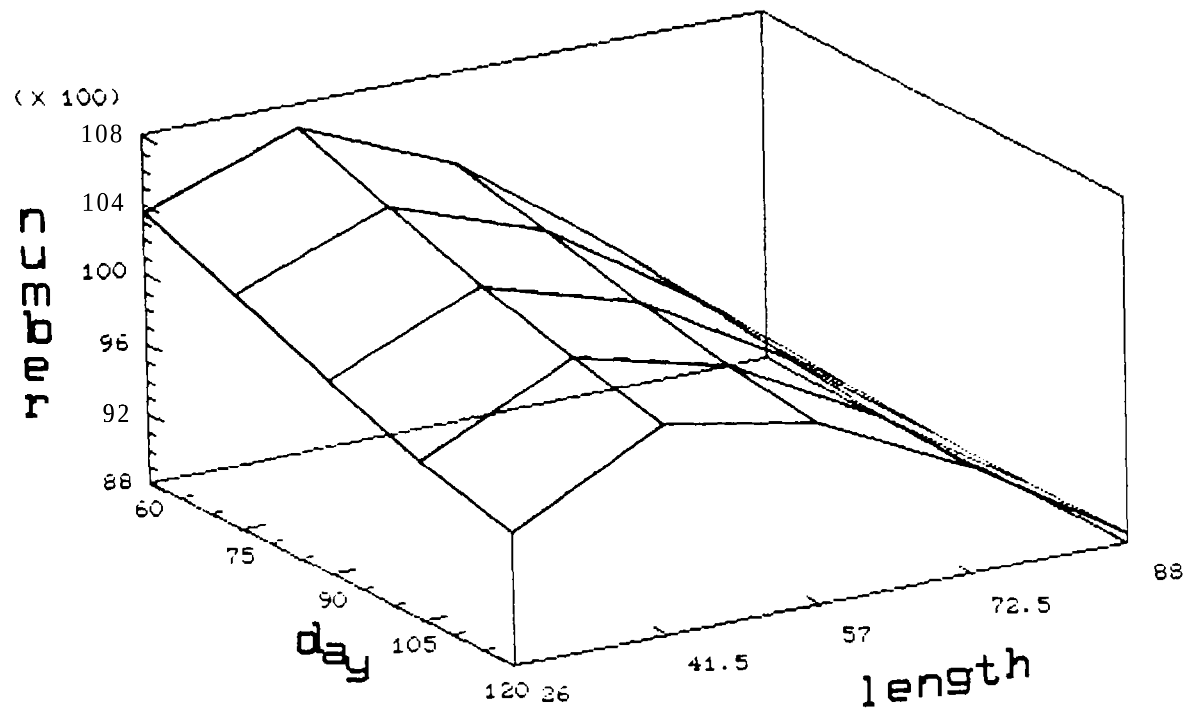


Figure 5.10 Response in Total Number of Ocean Residents (Number) at the End of the Simulation Period to Changes in Mean Arrival Date (Day) and Mean Length (Length) on Arrival at the Estuary, Assuming that Mortality in the Ocean is Independent of Fish Length.

There are some clear advantages to having precursor models. Construction of the example model began with a central hypothesis--that growth is the principal driving factor--and the model structure was built around this concept. The choice of growth as the principal factor was an arbitrary one, based on an interpretation of available literature. Other researchers might disagree with this view of the system. With a preliminary model, at least a focal point for debate exists which is sure to enrich future model development and understanding. Alternative models which assume that predation or physical factors drive the system could be developed if existing evidence warrants their construction. Models which would be applicable to other species will require distinctive structures which reflect species-specific differences. For example, a linkage between growth and early marine survival of coho salmon, which make little use of the Columbia River estuary, is not supported by empirical evidence (Fisher and Pearcy 1988). Upwelling and ocean temperatures seem to have a major influence on coho survival (Nickelson 1986) though the mechanisms for this phenomenon are poorly understood.

The simulation results presented in the previous section provide an example of the type of information that a EEO models can provide. Knowing the shape of the response surface for key variables such as percent survival would be extremely beneficial to fishery managers trying to maximize returns from hatchery releases. Given the level of uncertainty surrounding the system, it is unlikely that a model can be constructed in the near future that will be able to predict these response surfaces with a reasonable degree of accuracy. However, a model can indicate the shape of the response and suggest adaptive management strategies or experiments that would provide useful information. Models also provide a tool which can be used to help understand past experiences.

Although models such as the EEO model presented here appear to raise more questions than they answer, models actually act to focus questions. Since a good model is based on clear-ly stated assumptions, usually representing "convent ional wisdom", the focused questions identify what are important data to gather. These data then allow the model to be tested--which also is a test of the assumptions. Deriving answers to questions in the real world is an expensive proposition. A reliable model might serve as a screening device to help determine research priorities by identifying key parameters and assumptions that have significant implications. For example, based on the simulation results discussed above, the question of size-dependent mortality in the ocean is a critical assumption that is worthy of further elucidation. Being able to justify this assumption requires hard evidence. Simulation can complement but never replace experimentation. As Grant (1986) notes, "Answers to real-world questions ultimately must be found in the real world."

A model with a large number of interacting parameters and variables provides fertile ground for hypothesis generation. In the example application provided, only two major questions were asked: what is the effect of changing length and timing of estuary arrivals on terminal variables, and how does the impact of these parameters change when the assumption of length-dependent survival in the ocean is removed? Many others could have been posed. For example: What impact will markedly changing the time series of growth coefficients have on the results? What happens if the estuary survival rates are changed to reflect varying smolt quality at the same time that length and timing of estuary arrivals are changing? How sensitive are the results to the value used for the critical migration length? These and many other similar questions could be addressed with the model.

A RESEARCH STRATEGY FOR UNDERSTANDING OF EARLY MARINE SURVIVAL

The evidence for a strong relationship between early marine survival of salmonids and estuarine and near-shore oceanic environmental conditions is substantial and convincing. However, the effect of the marine environment cannot be viewed in isolation. As Mahnken et al. (1984) note:

> Many salmon biologists have held the opinion that the most important factors determining salmonid survival are of the freshwater environment. It has become clear that the marine environment is at least as important. However, it would be unwise to trade one narrow view for another by focusing solely on the ocean as the determining factor in salmonid production. The ultimate ocean survival of salmonids is the result of the interaction of marine conditions with the physiological state of the fish, which was predetermined by its freshwater

> experience. Therefore, it is necessary that research on the ocean survival of salmonids include the developmental history and smolt quality of evaluated populations.

Yearly variations in marine environmental conditions will continue to influence the relative success of mitigation and enhancement measures within the Columbia which are designed to increase adult run size. Though nothing can be done to influence large-scale physical processes such as upwelling, there are valid reasons for including such processes within the domain of applicable research. Better understanding of early marine processes offers numerous opportunities for enhancing the success of the Fish and Wildlife Program. For example, Nickelson (1986) demonstrates that the relative marine survival of wild coho smolts is nearly twice that of hatchery-reared smolts in years of weak upwelling, while survival of the two groups is equivalent in strong upwelling years. The obvious question is why does upwelling have such a powerful influence on hatchery stocks and not on wild stocks? If one could find ways of improving the early ocean survival of hatchery stocks in weak upwelling years such that it approached that of wild stocks, substantial increases in adult production could be realized. Assuming that weak upwelling occurs at the same frequency as strong upwelling (since 1958, there have actually been more years of weak upwelling off the coast of Oregon than years of strong upwelling), raising the survival rate of hatchery stocks to that of wild stocks would be expected to increase adult production of hatchery coho salmon by 33 percent.

Research Needs

The major dilemma at this point is how to enhance the early marine survival of hatchery-raised fish. This is not a new question nor is the answer easy. Since 1983, there have been three noteworthy instances in which efforts have been made to identify high-priority research which is needed to address this issue. In 1983 the Cooperative Institute for Marine Resources Studies sponsored a workshop on “the influence of ocean conditions on the production of salmonids in the North Pacific” (Pear-cy 1984). Work groups from this workshop produced recommendations for research on salmonid growth and survival in four areas: coastal regions, estuaries and inlets, hatcheries, and oceanic areas. A similar workshop in

1985 was sponsored by BPA and addressed issues in "improving hatchery effectiveness as related to smoltification" (Bouck 1987). A final product of this workshop was a list of twenty-one recommended research projects, ranked according to priority. This same topic has also been considered by the technical work group on improving hatchery effectiveness established by the Northwest Power Planning Council in 1987.

These collaborative efforts have provided valuable opportunities for discussion and synthesis among researchers and managers working on different aspects of the same general problem. Individuals who are highly qualified to identify research priorities have been involved in these efforts and we respect their judgement. The research recommendations of these work groups provide an excellent starting point from which to proceed to the more specific tasks of designing and implementing research projects intended to reduce the uncertainty surrounding early marine survival.

In their recommendations, the workshop participants and the technical work group consistently emphasize the need for better understanding of the relationship between smolt "quality" and marine survival. Four basic aspects of smolt quality are commonly mentioned:

(1) physiological readiness of the smolts to enter seawater,
(2) incidence of disease and other causes of undue stress,
(3) size of individual smolts at seawater entry, and
(4) timing of entry into the marine environment.

The general consensus is that understanding of the relative importance of each of these is contingent upon the development of indices which accurately measure (1) and (2) above, and an extensive and rigorous data collection effort which dutifully records all of the information necessary to quantify the above four factors, hatchery rearing conditions, relevant environmental factors, and smolt-to-adult survival for hatchery stocks within the Columbia Basin.

The Use of Models

Implicit in much of the research which has been proposed is the assumption that a comprehensive and systematic study of the impact of hatchery rearing conditions and release strategies on smolt-to-adult survival for each stock would result in a predictive model of survival. This model could then be used to design a rearing and release strategy that would enhance the chances of survival for each stock. The type of analysis that is suggested by this approach is along the lines of a multivariate, linear (or non-linear) regression analysis. Such an analysis would produce a statistical model with useful predictive properties from an immediate management perspective, but would have severe limitations as a long-range research planning tool.

There are important distinctions between what Caswell (1976) describes as models for prediction (usually derived through statistical analyses) and models for understanding. An example of a predictive model is a multiple regression model in which only those factors which explain a significant portion of the observed variation are included within the model. Predictive models can be quite accurate while having an underlying structure which bears little relationship to the phenomenon(a) being modeled. For example, one might construct a multiple regression model which expresses juvenile growth rate in hatcheries as a function of rearing density, water temperature, water flow, and nutritional measurements. By using a series of controlled experiments in which all of the factors are systematically varied, one might derive an empirical relationship which is quite valuable to the hatchery manager.

In such cases, it is relatively unimportant that a linear model (or even a nonlinear model) is at best a gross approximation of the complex relationships among the many factors that affect the output, as long as the models are suitably accurate for their intended purposes. However, an important constraint on the use of these models is that they apply only to the factors and the range of conditions that were included in the original experiments. They cannot be applied in situations where certain factors occur at levels outside the experimental range, or where additional factors are involved (e.g., in the example above, the use of supplemental oxygen would preclude use of the model).

In contrast to models for prediction, models for understanding are judged more by their ability to provide insights into how a system operates than the accuracy of their predictions. Developers of such models focus on the interactions among system components and hypothesize relationships based on ecological theory and observations. The example model which was developed for the estuary and early ocean life stanza is of this type. Models such as the EEO model can play a vital role in research planning. Whereas the goal of research task is to test hypotheses, theoretically sound models can assist in the planning process by anticipating observable phenomena which should result if a given hypothesis is true. In the strictest sense, observation cannot verify a hypothesis but it can refute certain hypotheses while corroborating others. The best corroboration of a hypothesis is the prediction of an event or observation that cannot be explained by alternative hypotheses.

Statistical models which are based on past or existing conditions in the Columbia Basin will be of limited utility for predicting the impacts on the fisheries of changes in the system which are expected to result from implementation of the Fish and Wildlife Program. The number of hatchery-reared smolts reaching the Columbia Estuary could increase dramatically in the next few years and there could be substantial changes in the species and stock composition, and in the physiological condition of these smolts. Questions about the ability of the estuary and coastal region to support increased numbers, and the future importance of interactions between stocks will arise. Research is needed to address these questions, and modeling should be an important component of that research. Statistical models derived through regression analysis may be important in the short term to improve the efficiency of existing hatcheries, but a more forward-looking, dynamic modeling process is needed to anticipate future changes.

A Process for Reducing Uncertainty

Reducing uncertainty within the Columbia Basin should be viewed as an iterative, six-step process in which theoretical models are used within the research process to suggest needed research (Figure 5.11). These six steps are described below.

Step 1 – Conceptualization. Existing information, models, and data are brought together and used to formulate alternative hypotheses about how the system operates.

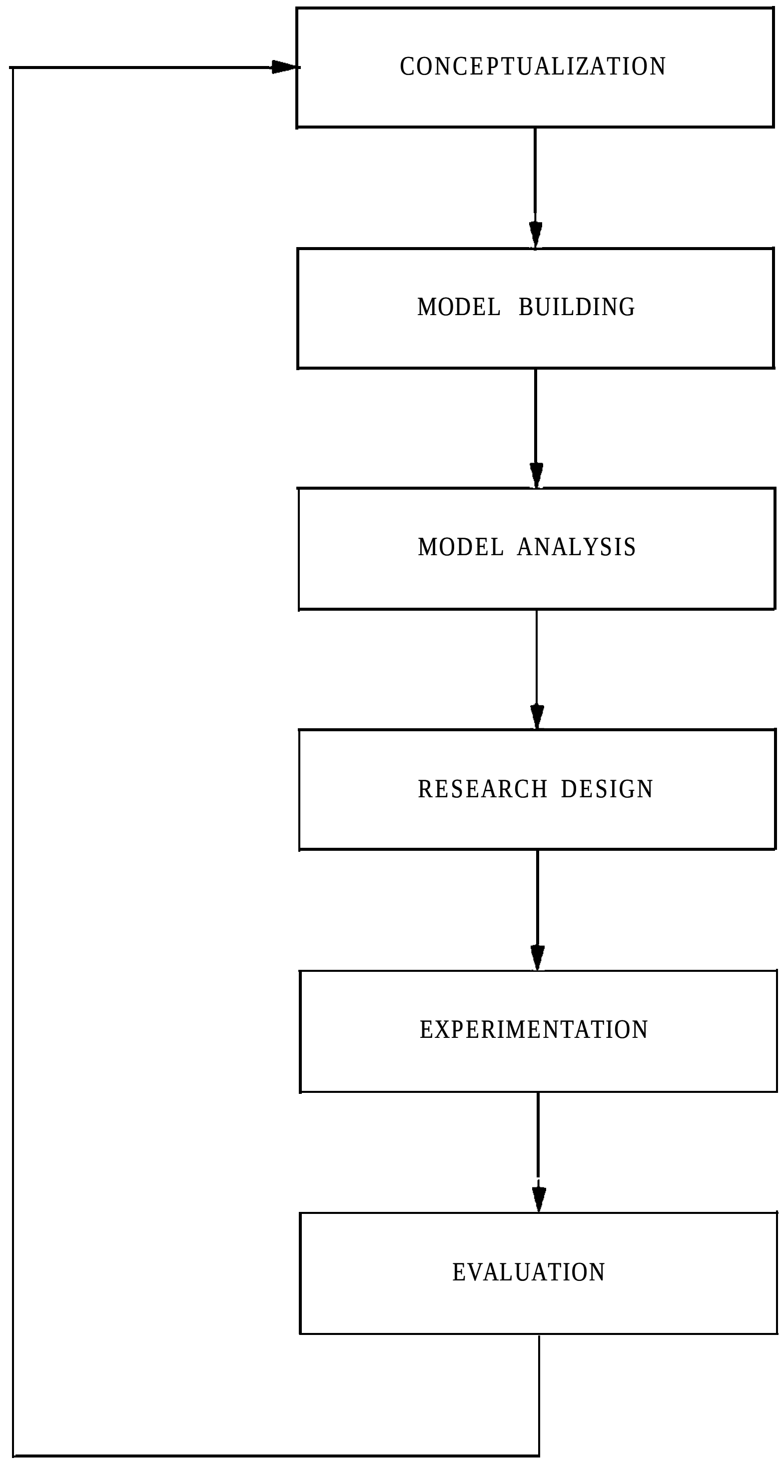


Figure 5.11 Flow Diagram of the Iterative Process for Increasing Understanding.

Step 2 Model Building. Alternative models are constructed which reflect the hypothesized relationships among system components. Existing data are used to estimate model parameters, as appropriate.

Step 3 – Model Analysis. Simulations under varied conditions and using alternative hypotheses are performed. The simulation results are compared and critical differences between models (in terms of model predictions) are identified.

Step 4 Research Design. Controlled experiments or adaptive management measures are planned, the results of which should refute at least one hypothesis.

Step 5 Experimentation. The planned research is executed.

Step 6 Evaluation. Experimental results are compared to model predictions leading to the rejection of certain hypotheses and to the corroboration of others. Return to Step 1.

In time, our understanding of the system should increase, the models should become more realistic in structure, and there should be a corresponding increase in the accuracy of model predictions.

The questions of who should maintain the models and who should plan the research is one which should be decided within the region. It is important that the organization given custody of the model and responsibility for the research have a long-standing commitment to increasing understanding of early marine processes. They should also be in the position of being able to plan and execute the necessary research. Technical work groups, while having the advantage of diverse representation from a variety of agencies, are probably not appropriate for long-term maintenance of such an effort. A more prudent approach may be to house the effort within one of the existing agencies such as the National Marine Fisheries Service and solicit input from other individuals and organizations on a regular basis. Considering the variety of agencies which have an interest in early marine survival of salmonids, opportunities for cooperative funding and research should exist.

CONCLUSIONS

The most promising research strategy for reducing the uncertainty concerning early marine survival is systemwide and comprehensive; it must be more than an *ad hoc* approach to a series of isolated problems. Indeed, the combination of modeling and research which is described above and illustrated in Figure 5.11 could be a productive approach to improved understanding of the entire range of biological questions involved in salmon and steelhead production. The key to this approach is to use research and management actions to test specific hypotheses, and to view modeling as a dynamic process that follows the learning curve. Collecting data in the hope that someone might someday be able to make sense of it, and building models based on hypotheses that can neither be refuted nor corroborated, are two classic ways of wasting valuable resources.

Chapter 6

MODELING THE LATE OCEAN LIFE STANZA

INTRODUCTION

Most of the research on the late ocean component of the salmon life cycle historically has focused on sport and commercial ocean harvest. Consistent with this view, a discussion of ocean harvest issues can be found in Part II of this volume. In addition, Part III outlines a mathematical programming approach to address the question of optimal allocation of fishing effort among several stocks, set within prescribed biological and administrative constraints. But allocation of harvest is not the only issue of importance in the ocean fisheries. Ecological issues such as growth rates, size structure, and age at sexual maturity interact with harvest rates to influence the relative productivity and fitness of a stock. These interactions can be conveniently explored with simulation models. As an example, this chapter describes a model which allows one to address interesting ecological questions in addition to harvest.

Two of the more important characteristics of a salmonid stock from a management perspective are age and size at adulthood, which are determined by growth rates and maturity schedules. Since fecundity generally increases with increasing age and size in salmon, these factors are important determinants of the reproductive potential of a stock. The reproductive potential is a vital component of the compensatory capacity of a stock to withstand harvest and environmental degradation. Growth rates and maturity schedules may also affect the economic value of the harvest. Salmon generally are sold by weight, and different species may be harvested at different stages in their life cycle. For example, pink, chum, and sockeye salmon are harvested as the fish are returning to their natal streams to spawn and growth is completed. In contrast, chinook and coho salmon are extensively harvested in the ocean during periods of rapid growth (Peterman 1985; Ricker 1981). Selective harvest pressures may alter the genetic composition of stocks of all species, but especially of chinook and coho salmon, by removing faster-growing or later--maturing fish from the gene pool. This can result in decreased mean age and size at maturity (Hankin and McKelvey 1985; Ricker 1981).

A number of authors have investigated the competitive advantages offered to salmon by various life history strategies or plasticity in age and size at sexual maturity. Schaffer and Elson (1975), Healey and Heard (1984, 1985), Stearns and Crandall (1984), van den Berghe and Gross (1984), Gross (1985), and Holtby and Healey (1986) discuss the evolutionary theories involved and provide species-specific applications. The general consensus of these authors is that variability or plasticity in age and size at maturity is advantageous for most populations. However, there are situations in which smaller, younger-maturing fish might have an advantage, or in alternative circumstances, larger, older-maturing fish might be positively selected. An example of the former is a situation in which there is an extensive ocean fishery which harvests a higher percentage of older-maturing fish than fish maturing earlier. Alternatively, stocks which require extensive energy reserves to make a prolonged spawning migration might be expected to have a higher percentage of larger, older fish successfully spawning. Apparently, age and size of maturity is not influenced only by genetic composition, but environmental conditions as well (Bilton et al. 1982; Peterman 1985). Conditions favorable to rapid growth during the early years of life seem to lead to earlier ages at maturity.

The example model presented here allows one to experiment with various combinations of growth rates, harvest rates, and mechanisms for determining the fraction of the population which matures each year (return schedules). Options are provided such that one can specify return schedules as a function of both age and size. Other models which have been applied to Columbia stocks (e.g. Johnson 1975, 1978; Webb et al. 1986) do not permit this flexibility. Output from the model includes age and length distribution of returning adults. This output can be combined with external information including length-fecundity relationships to produce estimates of reproductive potential.

MODEL CONCEPTUALIZATION AND STRUCTURE

Conceptually, the model follows a single cohort from January 1 of the calender year subsequent to marine entry until all members of the cohort have returned to the mouth of the Columbia or died (referred to here as the "late ocean" life stanza). The model includes components of fish growth, natural mortality, harvest mortality, and age and length at maturity. The first three of these processes are represented simplistically. Fish enter the model at a certain age (depending on their age at smolting) and with an assumed lognormal length distribution specified by the input parameters. Fish numbers and the length distribution are then modified in each time step. The initial time step refers to the period between January 1 and the mean date at which the youngest-maturing fish (usually jacks) return to their natal river. The length of this time step will depend on the life history characteristics of the stock in question. Subsequent time steps refer to a one year period between spawning migrations.

During each time step in the model, the number of fish remaining in the ocean is updated according to the equation:

$$N_{t+1} = N_t \times (1 - h_t) \times (1 - m_t) \times (1 - r_t), \qquad (6\text{-}1)$$

where N refers to number of fish remaining in the ocean, h, m and r refer to the probability of being harvested, dying of natural causes, or returning to spawn, respectively, in the time period t to t+l. Parameter vectors $\bar{h}$ and $\bar{m}$ are constants specified by the model user in the control parameter set. Values for r_t are derived in a more complicated fashion discussed below. A compensatory process is assumed such that:

$$\text{Number harvested } (H_t) = N_t \times h_t, \qquad (6\text{-}2)$$

$$\text{Number died } (M_t) = N_t \times (1 - h_t) \times m_t, \qquad (6\text{-}3)$$

$$\text{Number returning } (R_t) = N_t \times (1 - h_t) \times (1 - mt) \times r_t. \qquad (6\text{-}4)$$

The length distribution of fish remaining in the ocean is modified in a two step process representing the two mechanisms by which this distribution is changed, growth and emigration. Because fish length (L) is assumed to follow a lognormal distribution, as is appropriate when growth is proportionate to size (Boswell et al. 1979), the bulk of the calculations are performed using the natural log transformation of fish length (1). Prior to updating fish numbers, the expected value of the natural log of fish length (μ_l) is increased by the value of the growth coefficient (g_t). This represents a proportionate increase in length for all members of the population by a factor of $\exp\{g_t\}$. The growth coefficient vector ($\bar{g}$) is composed of constants specified in the control parameter set. Following the calculation of numbers and size of fish returning (discussed below), μ_l is adjusted for emigration using the equation:

$$\mu_{l,\,t+1} = ((N_t - H_t - M_t) \times \mu_{l,t}) - (R_t \times \mu_{r,t})) \;/\; (N_t - H_t - M_t), \quad (6\text{-}5)$$

where $\mu_{r,t}$ refers to the mean length of fish returning to spawn. This formulation assumes no size-selective mortality factors within age classes. The standard deviation of the natural log of fish length (σ_l), is assumed to remain constant over time. Though size-selective emigration will affect the variation in length, this relationship is not incorporated in the model because other factors omitted from the model may affect σ_l as well. Lacking any information on the sum effect of all factors on length variance, assuming a constant value for σ_l seems appropriate.

The numbers and length of maturing fish that return to the river at each time step is a function of the minimum maturing length ($l_{m,t}$) and the probability of maturing (p_t), given that an individual is larger than $l_{m,t}$. Generally, these parameters are specified in the control parameter set and are age-specific. The probability of return (r_t) is calculated as the product of the probability of being greater than $l_{m\,t}$ in length (endogenously determined) and p_t. An option in the'model allows one to simulate the special case where returning fish constitute the largest fish in the cohort. In this formulation, r_t equals p_t, and the value of $l_{m,t}$ is determined within the model such that the probability of being larger than $l_{m,t}$ is equal to p_t. The value of $\mu_{r,t}$, used in equation (6-5), is obtained by integrating $x \cdot f(x)$ over the range $l_{m,t}$ to $\{\mu_{l,t} + 3\sigma_l\}$ where f(x) refers to the normal probability density function. All length comparisons and calculations are made after length is adjusted for growth.

APPLICATION AND RESULTS

The model was applied to a single cohort from a fictitious spring chinook stock which migrates as yearling smolts and returns after spending 1 to 5 winters in the ocean. The purpose of this application is to explore the relative importance of selected mechanisms which affect the age and size structure of maturing adults and the potential implications for stock reproductive potential. For sake of simplicity, the initial cohort size was set at 100,000. Parameter values used in the model application (Table 6.1) were chosen such that they would be generally consistent with life history and harvest parameters given in the literature (e.g. Howell et al. 1985; Council 1986) but are not meant to represent any particular Columbia River stock. No attempt was made to verify the model through rigorous comparisons with empirical data.

Three scenarios were constructed to represent different life history strategies regarding age and length at maturity. In the first scenario (Case 1), it is assumed that the largest fish in each age class mature each year ($Prob(l > l_{m,t}) = p_t$). In the second scenario (Case 2), maturing fish are larger in size than other members of the cohort during the early years but become progressively more representative of the entire ocean population in later years (i.e., $Prob(l > l_{m,t}) = 0.16, 0.5, 0.75, 0.98, 0.99$; for t = 1 to 6, respectively). In the final scenario (Case 3), all but the very smallest fish in the cohort had an equal probability of maturing each year ($Prob(l > l_{m,t}) = 0.994$; for all t). Each scenario was simulated under two conditions, moderate ocean harvest and no harvest.

In order to evaluate the relationship between return schedules and reproductive potential, estimates of potential egg production for each simulated cohort were calculated. The potential number of eggs produced at each time step was estimated as the product of number of fish maturing, the fraction that are female (the values used were 0.2, 0.5, 0.6, 0.6, and 0.8 for t=1 to 5, respectively) and the average fecundity value calculated using the mean length of fish maturing in that time step and the length-fecundity equations given by Healey and Hear-d (1984) for Columbia River chinook salmon. Healey and Heard (1984) first convert fork length (FL) to

Table 6.1 Parameter Values Used in the Model Application for Each Scenario. See text for complete scenario and parameter descriptions.

Parameter[a]	Time step(t)	Scenario: Case 1	Case 2	Case 3
Initial number	0	100,000	100,000	100,000
σ_l	all	0.1811	0.1811	0.1811
h_t[b]	1	0	0	0
	2	0.14	0.14	0.14
	3-5	0.2	0.2	0.2
m_t	1-4	0.5	0.5	0.5
	5	0.95	0.95	0.95
p_t	1	0.01	0.063	0.01
	2	0.1	0.2	0.1
	3	0.6	0.8	0.6
	4	0.9	0.95	0.9
	5	1.0	1.0	1.0
g_t	1	0.43	0.43	0.43
	2	0.36	0.36	0.36
	3	0.31	0.31	0.31
	4	0.09	0.09	0.09
	5	0.01	0.01	0.01
$l_{m,t}$[c]	1	6.528	6.288	5.650
	2	6.694	6.467	6.014
	3	6.692	6.654	6.324
	4	6.422	6.505	6.414
	5	6.422	6.330	6.424

[a] Parameters: standard deviation of transformed length (σ_l), prob. of harvest (h_t), natural mortality prob. (m_t), prob. of maturing (p_t), growth coefficient (g_t), and minimum transformed length at maturity ($l_{m,t}$).

[b] Harvest rates in effect during ocean harvest conditions.

[c] Calculated internally by model in Case 1, user-defined otherwise.

postorbit-hypural length (POH) according to the equation: POH = 0.761FL + 47.2. Fecundity is then calculated according to the log-linear relationship: $\log_e(\text{fecundity}) = 1.754[\log_e(\text{POH})] - 2.95$. This relationship is based on data reported by Galbreath and Ridenhour (1964); fecundity estimates based on the linear relationship reported in Galbreath and Ridenhour (1964) will not substantially differ from those derived here. Summing the potential egg production over all time steps gave the total reproductive potential (unadjusted) for the simulated stock.

In order to simulate environmental conditions that might favor larger fish (e.g., a demanding upstream migration) an index of relative fitness (F_t) was introduced based on fish length. This index was standardized such that fish with the highest mean length at return in the exercise (denoted $\mu_{r,max}$), would have an index value of unity; all others would have an index value proportionately smaller (i.e., $F_t = \mu_{r,t} / \mu_{r\ max}$). This factor was incorporated in the product determining potential egg production, discussed above, and following summation produced an adjusted estimate of total egg production.

The parameter values for p_t and $l_{m,t}$ used in the model were deliberately chosen such that there would be little variation in numbers and age distributions among results from the three scenarios. This permits one to focus on the impact of harvest and varying lengths at maturity without the confounding problem of differences in age at maturity. The results suggest that harvest reduces the number of returning adults and shifts the age distribution of fish in the ocean and of returning adults towards younger fish (Table 6.2). This occurs because older fish are exposed to higher levels of cumulative harvest in the model. Also, because of the compensatory nature in which harvest is incorporated in the model, harvest totals are much greater than the difference between the number of returning adults under harvest and no harvest conditions. For example, under the ocean harvest conditions specified, roughly 11% of the fictional cohort were harvested, 8% returned to the river mouth and 80% were lost to natural mortality averaged across all scenarios. With no harvest, the percentage returning increased by three percentage points. The three scenarios exhibit the most pronounced differences among them in the mean lengths of returning adults and of fish remaining at sea (Figure 6.1). In

Table 6.2 Model Numeric Output Based on an Initial Population of 100,000 Individuals. Values in the table refer to numbers of fish under specified conditions. See text for complete description of model application.

		No Ocean Harvest		Modest Ocean Harvest		
Case	Age[a]	Remaining in ocean	Returning to spawn	Remaining in ocean	Returning to spawn	Harvest
1	3	49,500	500	49,500	500	0
	4	22,275	2,475	19,156	2,129	6,930
	5	4,455	6,683	3,065	4,598	3,831
	6	223	2,005	123	1,103	613
	7	0	11	0	5	25
Total			11,674		8,335	11,399
2	4	49,504	476	49,504	476	0
	5	22,313	2,439	19,189	2,097	6,931
		4,762	6,395	3,376	4,400	3,838
	6	352	2,028	194	1,116	655
	7	0	15	0	7	39
Total			11,353		8,089	11,463
3	3	49,504	496	49,504	496	0
	4	22,295	2,457	19,174	2,113	6,931
	5	4,510	6,638	3,103	4,567	3,835
	6	242	2,013	133	1,108	621
	7	0	12	0	5	27
Total			11,616		8,289	11,414

[a]Refers to year of life. All fish are assumed to enter the ocean during their second year.

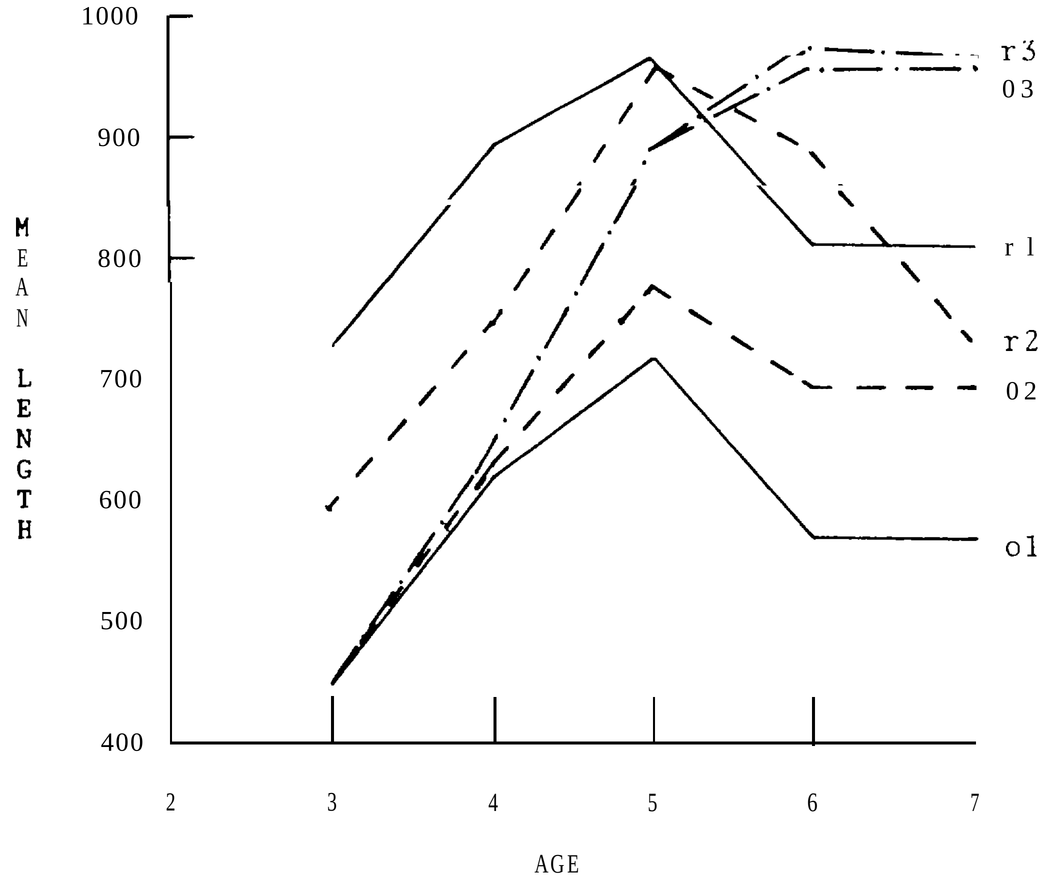


Figure 6.1 Mean Lengths (in mm) of Fish Remaining in Ocean (o) and of Maturing Adults (r) with each Scenario (1, 2, or 3) Plotted Versus Year of Life (age).

Case 1 and Case 2, mean lengths in year 6 decreased from the peak length attained in year 5. This occurs because the larger members of the cohort have matured and returned to the river by the end of the fifth year of life and only the smaller, slower growing members (in absolute terms) of the cohort remain in the ocean. The increase in mean length due to growth during the remaining time periods is not sufficient to overcome the reduction in mean length due to emigration of the larger individuals. Since size-selective emigration is reduced in Case 3, the mean length at return begins at the lowest level observed and increases steadily until the final year. The mean lengths of fish remaining in the ocean and of returning adults exhibit similar patterns except that the relative ranking of the three scenarios is reversed. Case 3 consistently exhibits the largest ocean size, followed by Case 2 and Case 1. Harvest does not affect the length at return for individual age classes but does decrease the overall mean age and length at return. This occurs by shifting the age distribution towards younger, smaller individuals (without harvest, mean age = 3.88; with harvest, mean age = 3.76). As one might expect, the decrease in overall mean length at maturity is most pronounced in Case 3. Percent decreases in mean length at maturity were 0.1%, 1.3%, and 2.5% for Cases 1, 2, and 3, respectively.

Potential egg production (adjusted and unadjusted for size-dependent fitness) as calculated above were expressed in terms of reference to 1000 maturing adults (both sexes combined) for no-harvest conditions and for 1000 and 714 maturing adults when harvest occurred. The differences between reproductive potential estimates based on 1000 adults under harvest and no-harvest conditions reflect only the impact of reducing the mean age and length of maturity through harvest. Estimates based on 714 maturing adults include the reduction in numbers of returning adults observed under harvest conditions, combined with the effect of reducing the mean age and length at maturity. The estimated values of potential egg production were used to compute three indexes of relative reproductive potential. Index values reflect the reproductive potential under the conditions specified relative to a standard condition. Two indexes of comparative reproductive potential (CRP: unadjusted, and CRPA: adjusted for length-dependent fitness), were standardized such that under Case 3 and no harvest, these indices had a value of one. The third index, reproductive potential sensitivity (RPS), is a measure of population reproductive robustness and compares reproductive potential within life history strategies. It was standardized such that the unadjusted reproductive potential estimate for each case under conditions of no harvest had a value of one. The range of values exhibited by this index within each case reflects the relative sensitivity of a stock, in terms of reproductive potential, to factors influencing the age and size structure of mature adults.

For each index, calculated values were highest for Case 1, and followed in decreasing magnitude by Cases 2 and 3 (Table 6.3). The relative advantage of having the larger members of each cohort mature earlier (or alternatively stated, having the slower growing members mature later) increased when length-dependent fitness was included. CRPA values were greater than CRP values for Cases 1 and 2, but not for Case 3; the largest difference between related index values occurred between Case 1 and Case 3 when both harvest and size-dependent fitness were in effect (1.147 - 0.939 = 0.208).

DISCUSSION

An analysis of the indices presented in Table 4 suggests that a life history strategy in which the larger fish of each cohort mature each season offers several competitive advantages under specific conditions. To review, the assumptions inherent in the model include: (1) harvest is age-dependent but not size-dependent; (‘2) fecundity is size-dependent but not age-dependent; (3) the *a priori* probability of maturing at a given age is fixed; and (4) sex-ratios of maturing adults by age are fixed. Under these conditions, having size-dependent maturation schedules results in a higher absolute reproductive potential and less sensitivity to cumulative mortality processes, such as harvest, which decrease the mean age at maturity. The assumption that fecundity is a monotonically increasing function of fish length is basic to the displayed model behavior. It has been observed that fish length, while significant, generally accounts for less than 50% of the total variation observed in fecundity of chinook salmon (Healey and Heard 1984). This suggests that simply equating relative reproductive success with fish length may have over-estimated the

Table 6.3 Relative Reproductive Potential of Fish Stock Under Each Scenario and Harvest Condition. See text for discussion of indexes.

Conditions	Basis[a]	CRP	CRPA	RPS unadj.	RPS adjusted
No Harvest					
Case 1	1,000	1.109	1.111	1.0	0.947
Case 2	1,000	1.067	1.099	1.0	0.930
Case 3	1,000	1.0	1.0	1.0	0.903
Modest Harvest					
Case 1	1,000	1.092	1.147	0.983	0.932
Case 2	1,000	1.032	1.056	0.967	0.893
Case 3	1,000	0.952	0.939	0.952	0.848
Case 1	714	0.780	0.819	0.702	0.665
Case 2	714	0.737	0.754	0.690	0.638
Case 3	714	0.680	0.670	0.680	0.605

[a]Number of maturing adults (both sexes combined) on which the reproductive potential estimate is based.

importance of fish size, but is unlikely to have affected the relative ranking of the prescribed life history strategies.

It would be inappropriate to attach unwarranted significance to the implications suggested by this example model application. The scenarios developed here are intended to illustrate the utility of modeling the late ocean component in non-traditional ways and exploring various hypotheses concerning life history strategies and their management implications. Considerable refinement of model parameter values, comparison with empirical data, and a thorough discussion of model structure and results with reference to sound evolutionary theory are needed to adequately assess model integrity. However, if one accepts for the moment that the model captures a semblance of real-world truth, then interesting management implications emerge.

Consider a situation, such as might exist in the Columbia River Basin, where a management agency is trying to rebuild a severely depressed chinook salmon population in an upriver tributary. Improvements have been made such that there is plenty of adequate spawning and rearing habitat available but a scarcity of spawning adults inhibits the rebuilding effort. A decision is made to use outplanting of juveniles to bolster production and the management agency is in search of a suitable parental stock, spawning stock native to the area being unavailable. The model results discussed above would suggest that a favorable characteristic of the parental stock would be that the larger members of each cohort mature earlier. Such stocks should be distinguishable by the relative shape of the mean length versus age at maturity curve (e.g., Figure 6.1). Having a higher relative reproductive potential, these stocks should be able to colonize the area at a faster rate. Naturally, other factors involved in outplanting would be considered as well.

CONCLUSIONS AND FUTURE RESEARCH NEEDS

The model, as currently presented, represents an investigative tool which can be used to explore hypotheses concerning important ecological mechanisms operating in the late ocean component of the anadromous salmonid life cycle. The model has the potential to be useful in future management of Columbia River Basin salmon and steelhead stocks. Because it easily fits within the simulation structure being developed to represent the life cycle of anadromous salmonids, and produces results with potential management implications, it should be included among those components receiving further investigation and refinement.

The model can and should be modified to suit the needs of the researcher or manager interested in its use. Those interested in improving the accuracy of model predictions will find it necessary to expend the majority of effort towards refining model parameter estimates such that model behavior adequately mimics empirical data. Application to a particular stock will require information on harvest rates, natural mortality, and age-length relationships of fish in the ocean and of returning adults. Information on harvest rates, age, and length should be available for certain stocks from mark-recovery data, data from hatchery returns, and spawning area surveys. Accurate estimates of natural mortality rates will be difficult to obtain due to a variety of factors (see Furnell and Brett 1986; Ricker 1976). Model predictions will be sensitive to prescribed natural mortality values. One might find it convenient to constrain the model such that a desired age distribution is produced for a given set of harvest, growth, and maturity parameters and treat one or more of the natural mortality parameters as variables. One can then compare natural mortality estimates obtained in this fashion with those obtained independently as a check on the integrity of the model and the defined parameter set.

Those individuals who are more interested in theoretical applications will find the flexibility of the model useful and may even wish to alter model structure to allow greater flexibility. One such alteration would be an expansion to permit tracking of individual sexes. There is evidence to suggest that males and females may not be subjected to identical selective pressures in regard to length at maturity (Gross 1985; Holtby and Healey 1986); it might be interesting to include sex-dependent mechanisms in the model. While theoretical application would profit from improvements in model parameter estimates, the utility of such applications is not overtly dependent on accurate parameter estimates since comparisons generally are made in relative rather than absolute terms.

Chapter 7

MODELING THE UPSTREAM MIGRATION OF ADULT SALMONIDS

INTRODUCTION

Hydroelectric development of the Columbia River Basin poses major obstacles for adult Pacific salmon and anadromous trout trying to return to their natal spawning areas. The dams which clog the mainstem Columbia River and its major tributary, the Snake River, effectively block or impair the upstream migration of fish. Dams which are not equipped with fish passage facilities, such as Chief Joseph and Hells Canyon, prevent anadromous fish from using large areas where natural spawning occurred historically. Downstream, run-of-river dams equipped with fish ladders have less obvious, but equally real, impacts. Poor design or inefficient operation of installed passage facilities can result in delay or mortality of migrating fish, thus reducing the numbers or fitness of adults reaching upstream spawning areas.

The potential impact of dams on migrating adult salmon is best understood in the context of the biological constraints imposed on these fish. Pacific salmon are semelparous, i.e., they mature and spawn only once before dying. As these fish return from the ocean to natal rivers, physiological changes occur which permit the fish to tolerate freshwater and also prepare them for spawning. Banks (1969) review of the literature on the upstream migration of adult salmonids raises the following considerations. Since Pacific salmon stop eating upon entering fresh water, the energy required for basic metabolic processes, reproduction, and locomotion must come entirely from bodily reserves. To be successful, spawners must make efficient use of the energy available to them. Idler and Clemens (1959) concluded that Fraser River sockeye salmon used over 90% of body fat reserves and, in females, up to 60% of total protein by the end of spawning. The metabolic fuels available to individual fish are obviously limited; it seems likely that some fish will not have sufficient energy reserves to successfully migrate and spawn under- demanding environmental conditions (Brett 1962).

In their study of ten species (15 stocks total) of anadromous fish, Bernatchez et al. (1987) observed that the energetic efficiency of migration is positively correlated with distance traveled. Among the examined stocks, Columbia River chinook stocks exhibited the lowest energetic cost per distance traveled. They proposed that the difficulty of a long migration selects for larger, more energy-efficient individuals, and for those which orient more accurately. Williams and Brett (1987) suggested that under certain conditions the Fraser River and Thompson River canyons might hinder the passage of weaker swimmers, thus selecting for stronger swimmers among pink salmon. They also noted that the swimming performance of pink salmon deteriorates as maturation progresses.

While it is possible to calculate the total energy expended by fish to reach spawning areas and spawn (using estimates of caloric content), one cannot determine empirically to what extent dam passage and hydrosystem operations, separable from other factors, have contributed to energy demand. One hypothesis is that if the migration is made less arduous or time-consuming, then less energy will be required for locomotion; more energy will be available for reproduction; and the total reproductive success of the population should be enhanced. Improving passage of upstream migrants in the mainstem Columbia and Snake Rivers is likely to be an expensive undertaking. Models are needed which can demonstrate potential benefits and cost-effectiveness of passage improvements prior to implementation.

Improving upstream fish passage around obstacles presents a challenging engineering problem. What appears to be a simple question of how to construct and operate a passageway that will permit fish to move from Point A to Point B is made complicated by the complexities of fish behavior and biomechanics. Addressing this problem requires knowledge of fishway hydrology, fish behavior, and the bioenergetics of salmon ascending the fishway. The recent work of Orsborn and colleagues (Aaserude and Orsborn 1986; Orsborn 1986; Orsborn and Powers 1986; Powers and Orsborn 1986) and references contained therein provide a thorough discussion of this topic. Mechanisms for improving passage are outside the scope of this report.

In order to build a realistic and comprehensive model of upstream migration, one must understand the migratory behavior of individual fish. It seems reasonable to assume, based on evolutionary theory, that individuals will attempt to maximize their reproductive potential at the spawning grounds. For most species in the Columbia, successful spawning requires efficient use of energy reserves within the temporal and spatial constraints imposed by the necessity of covering relatively long distances within finite time periods. Delay, fatigue, and multiple dam crossings, made worse when fallback occurs, add to the total amount of time spent and distance traveled in migration. The combination of time spent and distance traveled in migration is hereinafter referred to as migration path. Extensive migration paths increase the energy required for swimming and likely frustrate the migration strategies of many fish. In order to assess the chances of an individual making a successful spawning migration, one must estimate the likelihood of available energy reserves being sufficient to cover the energetic demands of the migration. Thus arises the three necessary parts of an upstream migration model: (1) an estimate of the variability in the migration paths (both time and distance) experienced by a stock; (2) an estimate of the caloric demand associated with each migration path; and (3) integration of (1) and (2) to provide an estimate of the energy reserves available for spawning within a stock and the distribution of those reserves among individuals.

MODEL CONCEPTUALIZATION AND STRUCTURE

The example model presented here is designed to allow examination of the magnitude and variability of increases in migration paths which are associated with dam passage. An example is provided of an application of the model to a fictitious stock which must ascend four run-of-river dams to reach the spawning grounds. Apparently, there are no existing models which consider an entire migrating population from a major river system in comparable detail. On a more limited scale, sophisticated models of fish locomotion and hydrodynamics under controlled conditions have been reported (see Orsborn and Powers (1986), Webb (1974) and accompaning references). Calculation of caloric demand and bioenergetic potential can be complicated as the reports by Orsborn et al., Webb, and others attest. However, simple approximations of energetic relationships are used in the model presented here for the purpose of illustration.

The present example focuses on the delay and mortality associated with dam passage in the immediate vicinity of each dam. Factors within each reservoir which affect passage have not been incorporated in the example. Two such factors are harvest and flow rate. Existing operational constraints on the hydrosystem may preclude management actions which might have a measurable effect on adult migration rates through reservoirs, though a weak relationship between flow regime and the average time in migration has been demonstrated Osborne (1960). A possible exception is the prevention of excessively low flow which has been shown to delay migration of steelhead trout and chinook salmon in the Snake River (Liscom et al. 1985). Above these extremely low flow levels, the general consensus is that travel time increases with increasing flow.

Harvest is omitted from the example model, since the present focus is on hydropower impacts on migrating fish. Harvest will obviously reduce survival and may affect migration paths, though data are lacking which could provide insight concerning a relationship between harvest and migration paths. Under the assumption that all fish are equally susceptible to fishing and that fishing does not affect the energy reserves of fish which avoid harvest, the impact of harvest will be a simple reduction in numbers. If either of these assumptions are incorrect, then the influence of harvest on the reproductive capacity of a given stock is more complicated.

A fundamental relationship exists between system survival and the number of dam crossings required. If the probability of surviving each dam crossing is equal to p, the probability of a successful migration involving X dam crossings will be equal to p^X. Since X is a random variable, knowledge of the distribution of X is necessary to accurately assess system survivability at the population level. Using analytical methods, one can calculate expected probabilities that can serve as points of reference for simulation models. In the general case, where p may vary between dams, the expected probability of completing the migration, $E\{P\}$, is given by the equation

$$E\{P\} = \sum_{i=1}^{k} \left[\sum_{j=0}^{\infty} f_i^{\,j} \cdot (1 - f_i) \cdot p_i^{(1 + j)} \right] , \qquad (7\text{-}1)$$

where k = number of dams and f_i is the probability of passing back downstream and having to reascend the dam (fallback) at dam i. The summation limit (∞) may be replaced with a finite integer which produces a suitable degree of accuracy in calculation. If p and f remain constant across all dams, X will follow a Pascal distribution and

$$E(P\} = \sum_{X=k}^{\infty} \binom{X - 1}{k - 1} \cdot (1 - f)^k \cdot f^{(X - k)} \cdot p^X . \qquad (7\text{-}2)$$

Under these conditions, an approximation to E{P} can be calculated as $p^{k/(1-f)}$. This estimator will always have a value that is less than E(P). If X follows a Pascal distribution then the number of additional dam crossings (X-k) will follow a negative binomial distribution. For a discussion of the relationship of the Pascal distribution to the more familiar negative binomial, see Boswell et al. (1979).

In the example model, individual dams are represented by independent simulation units. Outputs from each unit are used as inputs to an adjacent upstream unit. A multiple-dam system can be constructed by linking a number of dam simulation units together. Conceptually, individual fish face a series of independent Bernoulli trials at each dam with separate consequences depending on the outcome of each trial (Figure 7.1). Fish reaching a dam are required to locate and ascend the fishway. If they failed to ascend the fishway, the fish are lost from the population. Those that ascended the ladder require a variable period of time to find and negotiate the ladder. Upstream of the dam, fish face two trials in succession. The first reflects the possibility of dying due to injury incurred at the dam or due to exhaustion or disease exacerbated by dam passage. The second trial represents the chance of passing back downstream and having to repeat the process. Additional delay time is added to those that fall back. Those which survive the dam passage and do not fall back

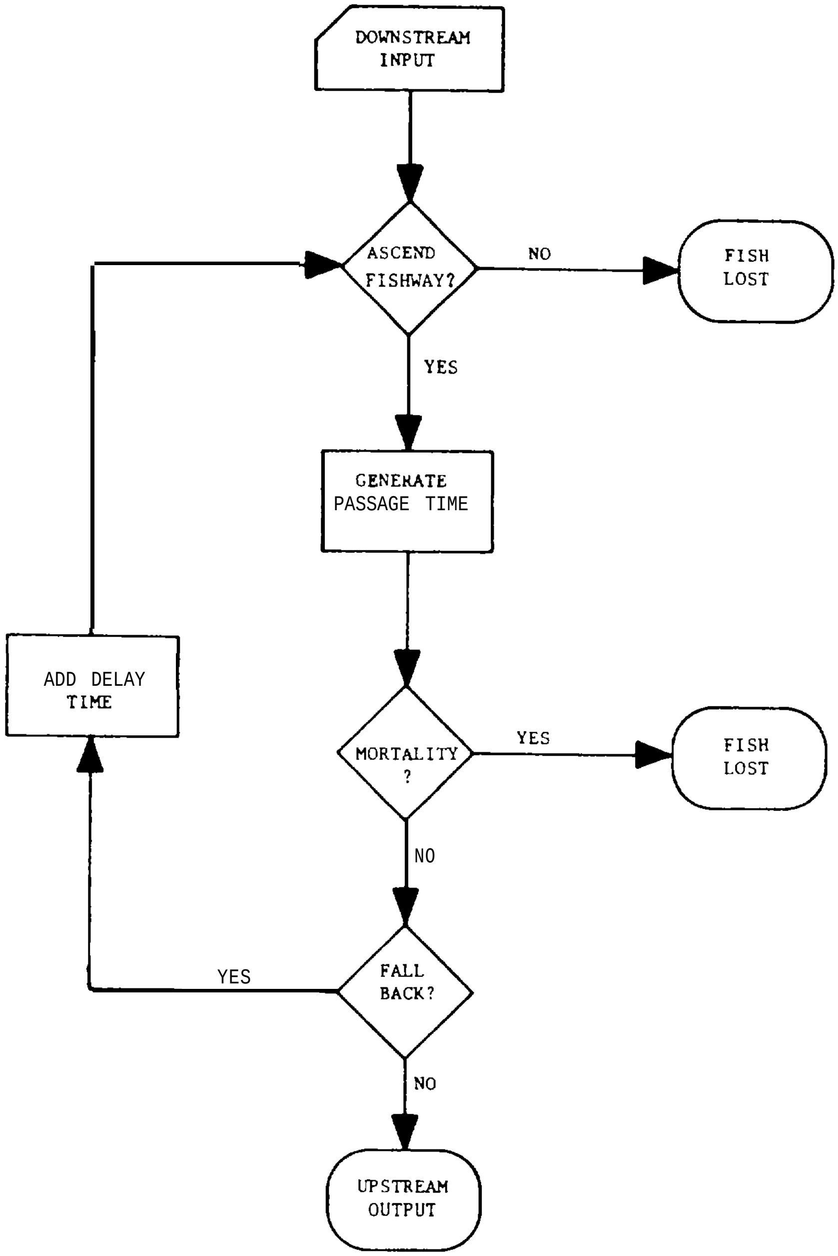


Figure 7.1 Conceptual Diagram of Upstream Passage Model.

proceed to the next dam. Because of fallback a fish may have to ascend a dam several times before moving upstream.

The model is designed to simulate the passage of individual fish through the system. Simulation of a migrating population involves a Monte Carlo approach in which multiple iterations of the model are run. For each iteration, the model tracks the migration of an individual through each successive dichotomy until an end point is reached (either the individual is lost or passes the last dam successfully). Probabilities are assigned for each Bernoulli trial in the control parameter set and the outcomes are then chosen randomly based on these probabilities. The cumulative time spent in the system, the number of times each dam is crossed, and the total number of dam crossings made are recorded for each simulated individual. Delay times associated with fallback remain constant as specified in the control parameter set. In contrast, the time in passage, T_i, is a random variable with probability density calculated according to the specified options, as explained below.

Two options are available within the model for defining T_i. In the first option, each dam crossing is treated as an independent event. Random passage times (in hours) were generated for each crossing, drawn from a three-parameter gamma probability distribution with shape, scale, and location parameters (α, λ, and Θ, respectively) specified in the control parameter set for each dam. Under the first option, the expected passage times for each crossing remain constant, regardless of the number of dam crossings made previously or the cumulative migration time at the time of crossing. The second option is designed to incorporate the effect of fatigue on the passage time of fish. Under this option, the gamma distribution parameters specified in the control parameter set are used as a reference standard. The standard expected time in passage for each dam, $E\{T_i\}$, is calculated as $E\{T_i\} = \Theta_i + \alpha_i/\lambda_i$. The scale parameter actually used to generate the passage time for each dam crossing, λ^*, is calculated as a function of the observed cumulative passage time (M) acquired up to the time of crossing and the expected prior cumulative passage time for the dam in question, where $E\{M_i\}$ equals $E\{T_1\} + E\{T_2\} + \ldots + E\{T_{(i-1)}\}$. This assumes an expectation of only one dam crossing per dam. The equation for

A* is

$$\lambda^{*} = \lambda_i \times (E\{M_i\}/M)^{\gamma_i}, \tag{7-3}$$

where γ_i, defined in the control parameter set for each dam, is an indicator of sensitivity to fatigue. This equation obviously does not apply to the first dam encountered if one begins with an expected and observed cumulative passage time of zero. The adjustment to λ in equation (7-3) will have the effect of decreasing the expected passage time of individuals that have been delayed less than expected, while increasing the expected passage time of individuals that have been excessively delayed and are assumed to be fatigued.

EXAMPLE APPLICATION

For purposes of illustration, the model was applied to a fictitious stock which migrates past a series of four dams. The dams were considered to be equivalent in terms of difficulty of passage. Four scenarios were constructed which allow comparison of the relative impacts of fallback and fatigue on migration paths and mortality. These scenarios involved the following combinations of fallback and fatigue: no fallback, no adjustment for fatigue (NFB-NF); no fallback, fatigue adjustment present (NFB-F); fallback present, no adjustment for fatigue (FB-NF); fallback present, fatigue adjustment present (FB-F). The control parameters for each dam were the same (Table 7.1). When applicable, the fatigue sensitivity factor, γ, was set at zero for the first dam encountered, increased to 0.5 for dams 2 and 3, and increased further to 1.0 for dam 4. This reflected a hypothesis that the ability of fish to recover from fatigue deteriorates as the migration proceeds. The result is an expectation of time in passage that becomes more sensitive to the ratio of expected to observed cumulative passage time as one proceeds upstream (Figure 7.2).

One thousand iterations of the model were run for each realization with one realization per scenario. Model output from iterations in which the simulated individual fish survived the migration were examined collectively within each realization. Distributions of total passage time (i.e., delay) and the total number of dam crossings when fallback occurred

Table 7.1 Control Parameters which are Constant for All Dams in the Model Application.

Parameter	Value
Probabilities: .	
below-dam survival	0.95
above-dam survival	0.95
fallback (when present)	0.20
Fallback delay time	4 hours
Delay time distribution:	
shape (α)	2
scale (X)	0.2
minimum (0)	3 hours

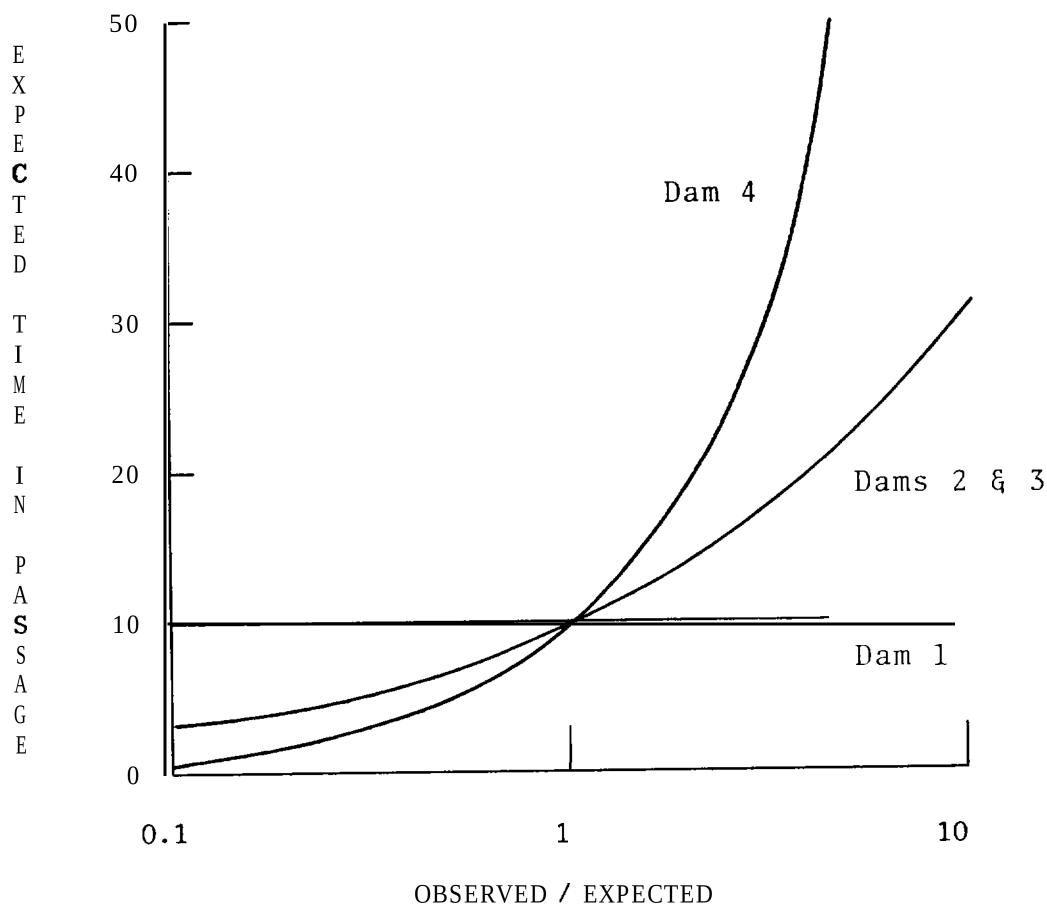


Figure 7.2 Expected Time in Passage (hrs) for Each Dam when Expressed as a Function of the Ratio of Cumulative Time in Passage (Observed) to Cumulative Expected Time in Passage (Expected).

were of primary interest. Probability distributions were fitted to output from each realization and comparisons made between scenarios.

In order to facilitate illustrative comparisons among scenarios, migration paths were converted into arbitrary units of bioenergetic demand. The basic metabolic unit (bmu) was defined as the amount of energy required by an individual fish to perform basic metabolic processes,-excluding swimming and reproduction, during migration for a period of 24 hours. The energy costs incurred due to delay and additional dam crossing were expressed using two different relationships which reflected alternative hypotheses of fish migration behavior. Under both hypotheses, it was assumed that fish normally migrate at the optimal rate in terms of energy efficiency, therefore, any increase in migration path created additional energy demand. Passing a dam in the upstream direction added an amount equal to the energy required to physically ascend the fishway (arbitrarily set at 3 bmu) in addition to energetic costs indirectly caused by delay.

Conceptually, there are two ways in which fish might respond to delay. First, they simply may proceed as usual with no change in swimming speed. Or, they might increase swimming speed in an effort to make up for lost time. Under the first hypothesis (Al), delay results in a linear increase in the total energetic cost of the migration, defined in the model as one bmu per day of delay. In contrast, if fish increase swimming speed in response to delay then the relationship between delay and energetic cost is curvilinear. Brett (1965) has demonstrated a U-shaped relationship between swimming speed of salmon and the energetic cost per unit distance traveled. If one assumes that migrating salmon would normally be swimming near the optimal speed and only increases in swimming speed are relevant, the descending portion of the curve can be ignored. The relationship between energetic cost and delay time as a second alternative (AZ) in this application was approximated using the equation:

$$\text{Cost(bmu)} = \exp\{0.015 \times \text{Delay(hrs)}\} - 1. \qquad (7\text{-}4)$$

This resulted in a curve which intersects the straight line representative of a linear relationship (Figure 7.3). Below the point of intersection, it is advantageous to increase swimming speed slightly to compensate for a short delay, but with longer delays, commensurable increases in speed will be energetically less efficient. The total energetic cost, C_t, due to

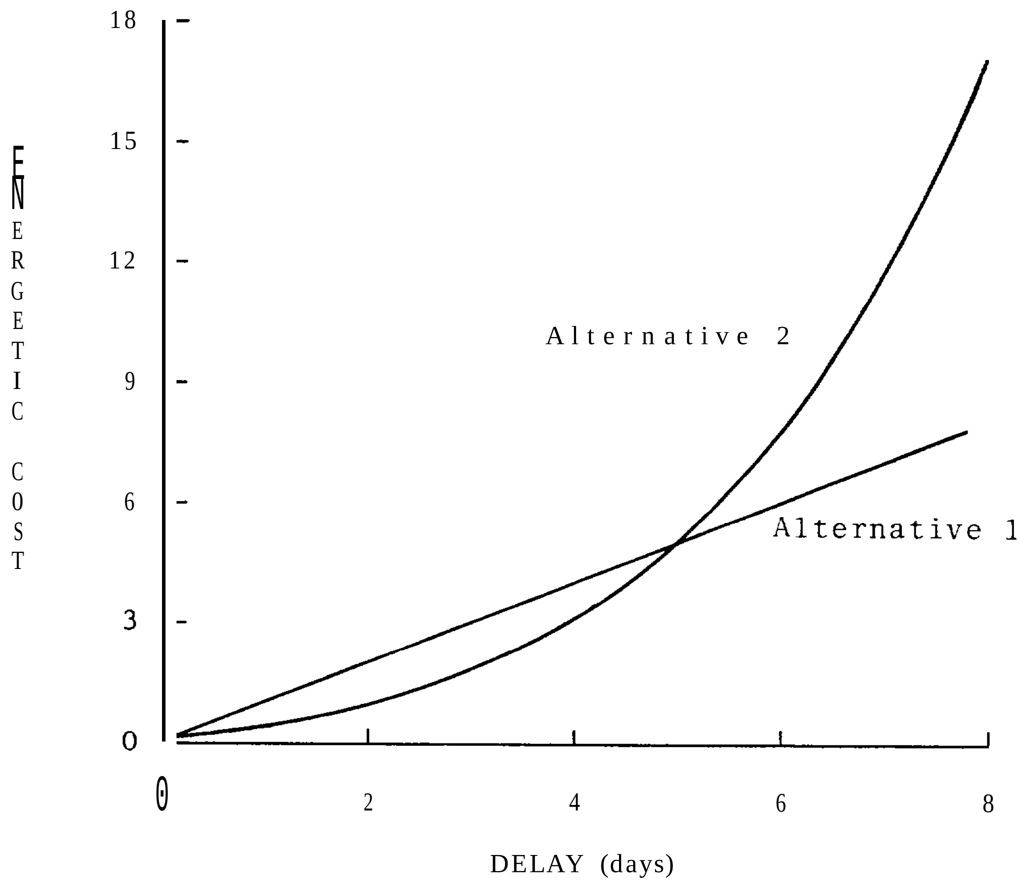


Figure 7.3 Additional Energetic Cost (bmu) Expressed as a Function of Delay Time Using Linear (Alternative 1) and Exponential (Alternative 2) Relationships.

delay and fallback is calculated as the sum of the cost due to re-ascending the dams (3 bmu x (Dam crossings - 4)) and the cost resulting from delay, calculated using the above relationships.

RESULTS AND DISCUSSION

The percentage of simulated individuals successfully completing the migration averaged 67% with no fallback present and 62% with fallback included. These values are within a few percentage points of the expected values (no fallback: 66.34%, with fallback: 60.3%) which were calculated using equation (7-2), suggesting no major bias in the simulation routine. A 7.6% decrease in survival that can be directly attributed to fallback was observed in the simulated system (expected value = 9.1%). Of those completing the migration (survivors) in scenarios including fallback, 55% made at least one additional dam crossing and 21% made three or more additional crossings. The average number of dam crossings made by survivors under fallback conditions was 4.86.

It is important to note that data from only survivors represents a censored data set that may give rise to misleading conclusions. As the probability of surviving a dam crossing decreases, the mean number of dam crossings made by system survivors is expected to decrease. This occurs because fish making additional dam crossings have a reduced probability of being sampled since they are more likely to be lost via mortality. To illustrate, if there had been no losses of fish in the simulated system, the expected number of dam crossings would be 5.0 ($E\{X\} = k/(1-f) = 5$), a value substantially higher than the mean number observed for survivors. Fitting a Pascal distribution to the data on total dam crossings of survivors produced an estimated probability of successful passage without falling back of 0.823, which is greater than the true value of 0.722. This type of bias, which may be unavoidable, must be accounted for in monitoring studies in which conclusions are based on information from a few individuals for which complete data records are available.

Since all survivors incurred a minimum passage time of 12 hours, reported times were adjusted downward by this amount to facilitate comparisons (Table 7.2). Probability distributions fit to the generated passage data provide a means of integrating information (Table 7.3). For Scenario NFB-NF, empirical fitting of a distribution was done only as a check as it was assumed that the data came from a gamma distribution with $\alpha = 8$ and $\lambda = 0.2$ based on statistical theory. Lognormal distributions were fit to passage time distributions for other scenarios as the lognormal consistently fit these data better than did the gamma or Weibull distributions, the only alternatives tested. All distributions were fit using Statgraphics™ routines (STSC 1986) and goodness-of-fit was evaluated using the Kolmogorov-Smirnov statistics provided by the software application.

Both fallback and the inclusion of the fatigue factor had a demonstrable impact on passage time of survivors (Figure 7.4). Inclusion of fatigue without fallback had the effect of skewing the distribution such that the median value was less than that for the reference case (NFB-NF), but there was also a higher percentage of individuals in the upper tail of the distribution. This resulted from a polarizing effect of fatigue in making short passage times shorter, while simultaneously increasing the longer passage times. Fallback without fatigue increased the mean, median, and variance of the passage times relative to the reference case. Inclusion of fallback and fatigue in combination had the most pronounced effect on the passage time distribution.

The relationships used in the provided illustration to calculate energetic costs associate a high energetic cost with passing a dam relative to that caused by delay. One additional dam crossing is equivalent to 3 days of delay in the linear model (Al) and up to 3.8 days of delay in the exponential model (A2); the actual value depends on prior delay time. As expected, the added energetic costs (costs) due to additional dam crossings and delay incurred when fallback was present were much larger than otherwise (Table 7.4). For this reason, further discussion of costs is limited to scenarios including fallback and focus on differences due to fatigue and alternative hypotheses concerning fish response to delay.

Table 7.2 Summary Statistics for Time in Passage (Hrs) of Simulated Populations Under Each Scenario. Scenario notation: NFB/FB (no fallback/fallback present), NF/F (no fatigue/fatigue factor present).

Statistic	Scenario NFB-NF	NFB-F	FB-NF	FB-F
Sample size	665	665	613	616
Mean	39.4	38.1	53.5	64.8
Median	38.0	33.9	48.9	51.4
Standard deviation	14.17	20.78	23.95	45.50
Standard error	0.55	0.81	0.97	1.83
Minimum	9.4	5.2	10.6	7.2
Maximum	106.5	166.6	159.9	316.2
Skewness	0.78	1.63	1.01	1.82

Table 7.3 Parameter Estimates and Kolmogorov-Smirnov Goodness-of-fit Statistics (D) for Probability Density Functions (Distribution) Fit to Time in Passage of Simulated Populations. All D values are not significant (approximate significance level equaled 0.999 in each case).

Scenario	Distribution[a]	Parameter[a]	Estimate	D
NFB-NF	Gamma	α	8.0	0.031
		β	0.2	
NFB-F	Lognormal	μ	3.505	0.026
		σ	0.5298	
FB-NF	Lognormal	μ	3.882	0.027
		σ	0.4507	
FB-F	Lognormal	μ	3.956	0.018
		σ	0.6619	

[a] Distribution and parameter notation follows Mood et al. (1974).

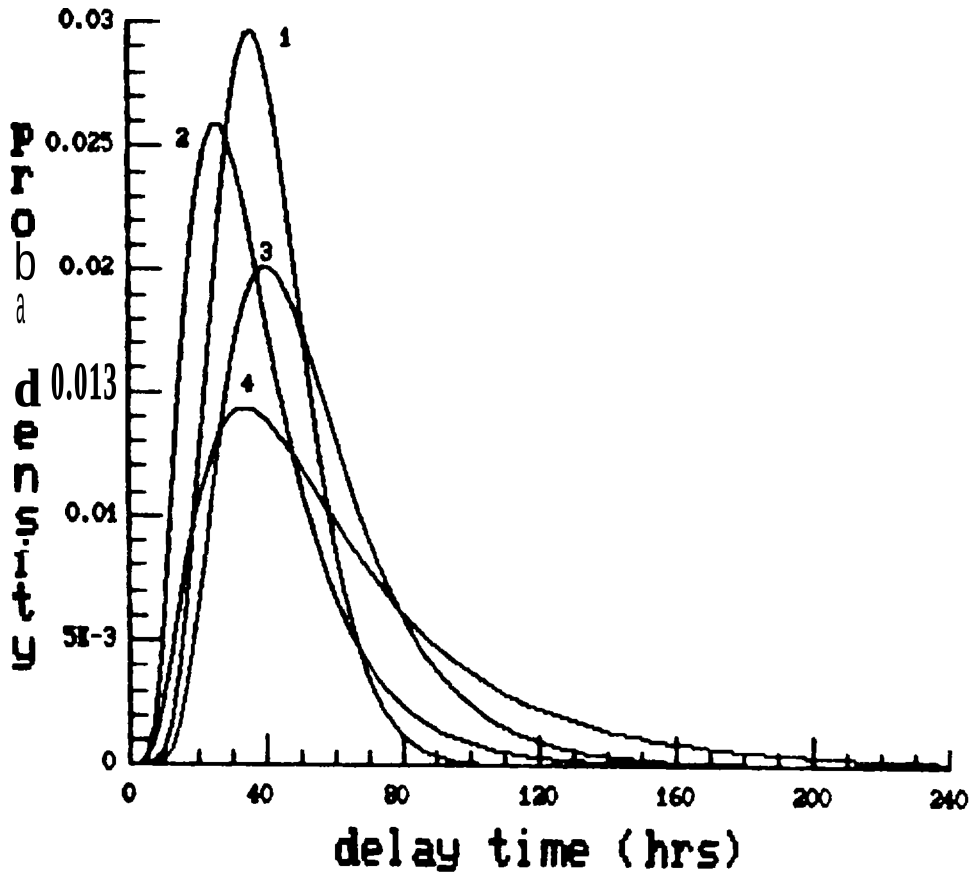


Figure 7.4 Probability Density Functions Fitted to Total Time in Passage (Hrs) for Scenarios NFB-NF (1), NFB-F (2), FB-NF (3), FB-F (4). See text for description of scenarios and distributions.

Table 7.4 Summary Statistics for Additional Energetic Cost (bmu) of Migration for Each of the Simulated Scenarios Using Linear (Alternative 1) and Exponential (Alternative 2) Cost-Delay Time Relationships.

Statistic	Scenario			
	NFB-NF	NFB-F	FB-NF	FB-F
Sample size	665	665	613	616
Alternative 1				
Mean	1.64	1.59	4.77	5.32
Median	1.59	1.41	4.60	4.64
Standard Deviation	0.59	0.87	3.80	4.72
Minimum	0.39	0.22	0.44	0.30
Maximum	4.44	6.94	19.57	32.88
Alternative 2				
Mean	1.85	1.88	4.95	6.50
Median	1.77	1.66	4.78	4.81
Standard Deviation	0.44	4.86	3.87	9.50
Minimum	1.15	1.08	1.17	1.11
Maximum	4.94	12.17	22.13	123.85

The cost estimates obtained from the simulations indicate an interaction between inclusion of the fatigue factor and the choice of relationships used to estimate the cost of delay. In Scenario FB-NF, the total cost estimated using the exponential model exceeded the estimate from the linear model by 177 bmu compared to a difference of 1,177 bmu in Scenario FB-F. Total estimated costs for Scenario FB-F exceeded those for Scenario FB-NF by 550 bmu and 1,556 bmu using Alternatives 1 and 2, respectively. These values reveal a synergetic relationship between fatigue and the exponential model that resulted in higher costs than expected based on an additive assumption: $C_t[1,1] + C_t[0,2] - C_t[0,1] << C_t[4,2]$. In the scenario notation introduced here, [i,j], i refers to the fatigue factor (O=absent, 1=present) and j denotes Al or A2. Fallback is assumed to be present.

A plot of the cumulative distribution of costs indicate that differences among scenarios are most evident in the upper tail of the distributions (Figure 7.5). The values plotted in Figure 5 are fairly close to one another for each of the four combinations until around the 60th percentile at which point [0,1] and [0,2] diverge from [1,1] and [1,2]. A second divergence occurs near the 85th percentile in which [1,1] separates from [1,2]. Little difference is observed between [0,1] and [0,2] throughout the range of plotted values.

Understanding the terminal impact of increased migration paths requires the translation of energy costs due to delay and additional dam crossings into population losses in reproductive potential at the spawning site. Conceptually, this might be accomplished in the following manner. Consider a population in which each individual has a certain amount of energy reserves available following a non-impeded migration. Assume that this available energy corresponds to an added migration cost of zero and a reproductive success level (RSL) of one on a scale from zero to one. Since added delay and dam crossings subtract from the available energy remaining at the conclusion of migration, RSL would be expected to decrease as the cost of migration increases. In the absence of empirical data, one might hypothesize various shapes for the relationship between RSL and energetic cost (Figure 7.6). A useful comparative measure is the average RSL value for the population. This can be calculated for the simulated populations

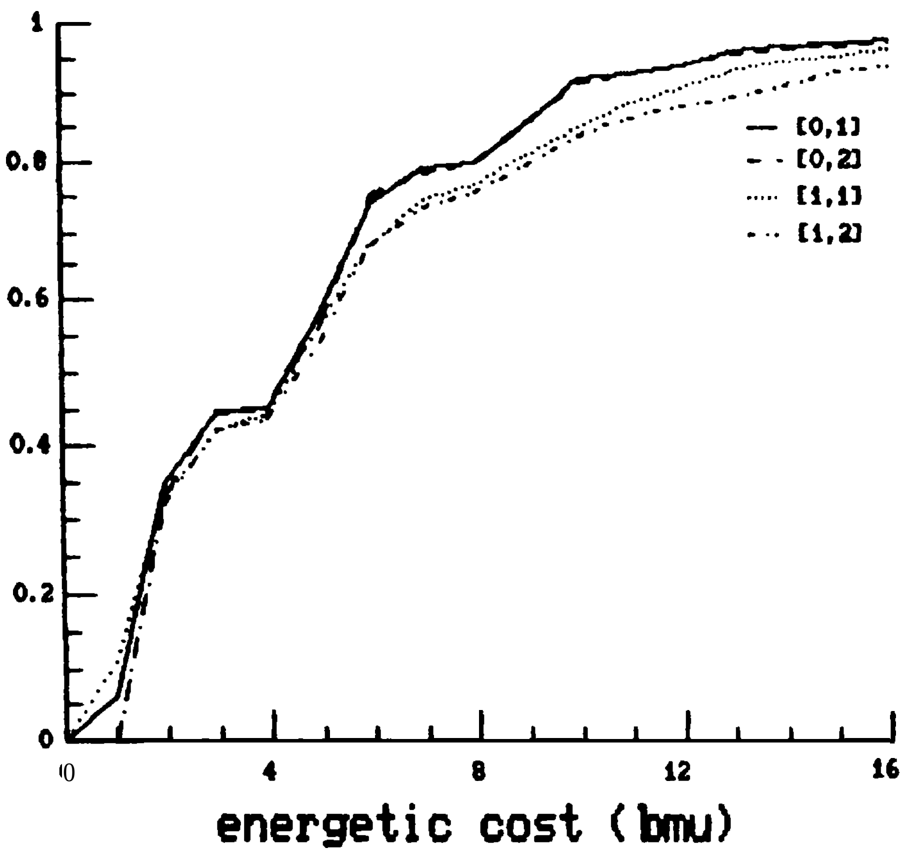


Figure 7.5 Cumulative Relative Frequency of Additional Energetic Costs Incurred in Migration for Scenarios Involving Fallback Under Two Methods of Assigning Costs to Delay Time. Scenario notation used: FB-NF-Al (0,1), FB-NF-A2 (0,2), FB-F-Al (1,1), FB-F-A2 (1,2). See text for description of scenarios.

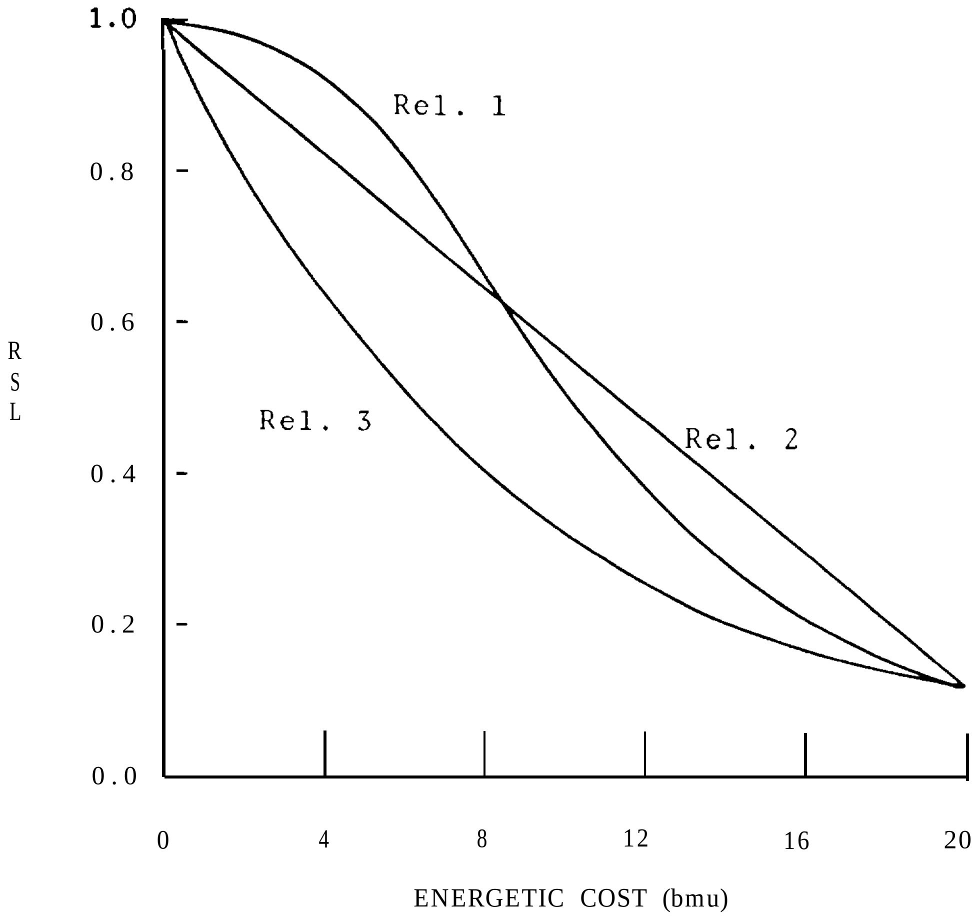


Figure 7.6 Reproductive Success Level (RSL) Plotted as a Function of Additional Energetic Cost Incurred In Migration for Three Hypothesized Relationships: Sigmoidal (Rel. 1), Linear (Rel. 2), and Exponential (Rel. 3).

as the sum of the individual RSL values divided by the number of survivors. An example of the application of this process to Scenarios FB-NF-Al and FB-F-A2 using the RSL-cost relationships depicted in Figure 7.6 is presented in Table 7.5. Clearly, population RSL values and the differences among them depend on which RSL-cost relationship is applied, in addition to being influenced by factors affecting estimates of cost.

CONCLUSIONS AND IMPLICATIONS FOR FUTURE MODELING

Based on the foregoing analysis, it seems clear that fallback, fatigue, and fish response to delay play significant roles in determining the extent of cumulative hydrosystem impacts on upstream migrants. The scenarios examined in the model application do not represent different systems or the same system operated in different ways, but rather are reflections of different conceptualizations of the same system. Each scenario represents a unique combination of assumptions concerning the underlying mechanisms influencing the system. The purpose of this exercise is to illustrate the importance of correctly portraying important system processes within a model used to evaluate management alternatives. As such, the insights to be gained from this investigation are more relevant to the task of modeling the system than operating it. Also, bear in mind that the relationships used in the model application were created _ad hoc_ and should not be interpreted as a reflection of any particular system.

The Monte Carlo approach used here provides an appropriate tool for simulating upstream migration and passage. This approach exhibits the flexibility needed to examine the system from an individual level while also allowing a population perspective. The model structure easily incorporates a level of complexity and resolution, combined with stochastic factors, that would be difficult to handle using alternative approaches. The model demonstrated that it could faithfully reproduce results predicted by analytical models while also including feedback mechanisms, such as the fatigue factor, that effectively would be intractable with an analytical approach.

Table 7.5 Average Reproductive Success Levels (RSL) Calculated with Three Alternative Relationships Between RSL and Added Energetic Cost Incurred in Migration. See text for description of scenarios and cost calculation.

	Reproductive Success Level[a]			
Scenario	RSL1	RSL2	RSL3	Range[b]
(a) FB-NF-Al	0.843	0.785	0.625	0.218
(b) FB-F-A2	0.794	0.707	0.585	0.209
(a) – (b)	0.049	0.078	0.040	
(b) / (a)	0.942	0.901	0.936	

[a] RSL values calculated as:

$$RSL1 = 1 - \left[\frac{cost^3}{cost^3 + 10^3} \right],$$

$$RSL2 = 1 - (0.045 \times cost),$$

$$RSL3 = \exp\{-0.1151 \times cost),$$

[b] Range is calculated as RSL1 - RSL3.

Fallback proved to be a vital factor in determining the impact of the hydroelectric system on upstream migrants. Relevant to system survival, changes in migration paths, and energetic costs incurred in migration, fallback exhibited a pronounced effect. The 20% fallback used in the simulations is equivalent to that observed by Liscom et al. (1985) for steelhead trout tagged and released above Lower Monumental Dam. Clearly, fallback is a distinct possibility for fish ascending mainstem Columbia and Snake River dams, especially during periods of heavy spill. It would be remiss to exclude this factor from any model constructed to simulate upstream passage.

While fatigue and the nature of fish response to delay were shown to play important roles conceptually, empirical evidence to substantiate and define these relationship likely will be difficult to obtain. Given these constraints, explicitly including these factors in a more specific model application than that presented here may not be practical. However, it may be possible to define model parameters to implicitly include some of the impact of these factors. Creative experimentation may also provide insights into their presence or absence and magnitude. Recognizing that such factors can exist and may have measurable impacts might be sufficient to appropriately temper interpretation of results from models lacking these features,

The process of converting changes in migration paths into additional energy demand and losses in reproductive potential proved to be a useful tactic in terms of placing these impacts in a proper context for comparison. As was alluded to earlier, methods are available in the literature to permit conversion of migration paths to caloric demand at a much finer level of resolution and accuracy than that used in this application. The potential gain from such an effort depends on the level of resolution needed to choose among management actions being considered. For example, a decision to alter fishway operations or design at a particular dam requires more detailed information than does a decision to fund a study on spillway operations to minimize fallback. Quanitative conversion of caloric demand into relative reproductive success with any reasonable level of certainty is beyond the present level of understanding of this topic. This should not preclude such attempts from being made in a conceptual modeling context. Further experimental work or closer inspection of existing data may provide insights into the nature of the RSL-cost relationship.

Chapter 8

SMOLT MONITORING IN THE COLUMBIA RIVER BASIN

INTRODUCTION

The monitoring or counting of Columbia River Basin juvenile salmon and steelhead trout (collectively referred to as smolts) as they migrate to the sea from hatcheries and natural nursery areas has two principal objectives:

(1) to keep track of the outmigration on a real-time basis in order to provide information to those responsible for manipulating streamflows and spills at dams, with the goal of increasing downstream passage survival; and

(2) to permit assessment of the biological and cost-effectiveness of the several mitigation policies under consideration.

If it were a matter of passing all the migrating smolts through a suitable counting device at each dam, the smolt monitoring problem would be relatively straightforward. One still would have to account for the interdam additions of fish and the mortality at and between dams, but an accurate count would allow tracking of the smolts. If fish managers had additional information from marking fish in hatcheries, and could make inferences about the sources of the counted fish (hatcheries vs. tributary streams vs. main-stem spawning habitat), they could design fairly straightforward experiments to explore survival response to various types and levels of mitigation measures.

The actual situation is not so simple. It is impossible to count all smolts at dams or anywhere else in the system, given current technologies and budgets. The volumes of water, sizes and numbers of fish, and the variety of routes past a dam (turbines, spillways, and bypass facilities) combine to make complete enumeration impossible. Researchers therefore have had to devise experimental methods that depend on sampling smolt populations. Since the accuracy of these sampling methods depend on many of the same variables that mitigation policies address (e.g., alteration of the size and quantity of flow), the sampling approach has increased the complexity of the research problem. Add to this difficulty the observation that environmental conditions change in uncontrollable, unpredictable, and even unobserved ways, and one begins to see why a respected student of the problem, Carl Walters, could say (Webb et al. 1986):

> It would be scientifically misleading and economically costly to pretend that the effects of any suite of time series experiments (in the form of independent sub-basin rehabilitation and monitoring programs) will eventually sort themselves out so as to provide a clear picture of which actions and strategies are actually performing as intended . . . There will continue to be misleading correlations between management measures, uncontrollable environmental changes, and patterns of survival and abundance.

This chapter will provide background on Walters' comment, discuss some technical issues that have assumed importance in the research literature on smolt monitoring, describe new technologies that may offer hope for remedies, and suggest possible directions for future research efforts.

THE FUNDAMENTALS OF MITIGATION POLICIES AND THEIR EVALUATION

The introduction of fish hatcheries offers to increase effective fish recruitment (additions to adult populations), particularly if spawning habitat has been destroyed or made inaccessible by siltation, pollution, or dams, or if losses of fish to the marine commercial fishery have threatened naturally occurring spawning runs. If inriver sport fisheries are important or, as in the case of the Columbia, if tribal fishing rights also must be respected, these hatcheries must be located far upstream. This is to ensure that returning adults will be available to upstream anglers.

In highly developed and controlled rivers such as the Columbia, upstream hatcheries imply that both upstream and downstream migrants will have to traverse man-made obstacles. Therefore, both wild and artificially spawned and raised juvenile fish will have a difficult time getting to the sea. In particular, dams with hydroelectric turbine installations may lie in the way. These create pools of relatively slack water with higher temperatures and perhaps larger predator populations (Raymond 1968; 1969). Although the literature is unclear on role of slack water on juvenile mortality, the Columbia system currently is being managed on the premise that increasing flows between dams will increase survival (Karr and Maher 1985).

However, the main threat to the safe passage of juvenile fish comes from the turbines. Fish passing through these may be injured or stunned, and thus made more vulnerable to predators, or killed outright. Several possibilities exist for the smolts to avoid the turbines. First, the smolts can use adult passage facilities (fish ladders) in reverse. This does not seem to work particularly well, probably because of the difficulty of attracting the fish into the upstream ends of such structures (Struthers 1984). An easier approach, which apparently enhances survival rates, is to spill water and let the smolts go over in the spillway. According to Schoeneman et al. (1961) mortality for passage over spillways is about 2 percent while that for turbines is about 11 percent. Although, spilling large volumes of water increases concentrations of gases in downstream waters (most importantly nitrogen), which can increase mortality rates in downstream waters due to gas bubble disease, physical modifications of spillways fortunately has mitigated this problem to a large degree. A third alternative is to provide specially designed and constructed smolt passage facilities, including fish guidance equipment, to keep the smolts out of the turbines and to encourage them to enter the bypass (Bently and Raymond 1969; Smith and Farr 1975). A goal of the regional fishery enhancement policy is to have such facilities at all dams on the Columbia.

A fourth alternative, transporting smolts by barge or tank truck from an upstream point to the estuary below Bonneville Dam, simultaneously avoids the hazards of both dams and slack water. Ebel (1980)) for example, reports that survival of transported fish is from 1.1 to 15 times higher than that of control fish which passed seven mainstem, run-of-river- dams and associated reservoirs. The success of recent transportation experiments have convinced many individuals in the Pacific Northwest that transportation should continue to play a major role in the Columbia Basin. If transportation were to become the sole mitigation alternative, the monitoring problem would look very different. Survival of the transpotted smolts could be measured directly, albeit at some cost, when the bar-ges or trucks were emptied. Survival to adulthood would be estimated by marking some fraction of the transported fish and observing their return rates. The bulk of the material contained in this report is concerned with the monitoring of fish which are not transported.

The monitoring of smolt migration may help to improve the effectiveness of such mitigation alternatives. For example, the recognition that a large number of smolts are actively migrating allows fish managers to target high stream flows for these periods. They can time spills to coincide with the arrival of smolts at the dams and with the hours when smolts are more likely to pass through the spillways. The evaluation of mitigation measures also depends on the monitoring of smolt migration. Since all mitigation measures cost money--transportation of smolts and fish passage facilities require capital expenditures, and involve operation, maintenance, and replacement costs, and spills and enhanced flows provide water for fish rather than for the potential generation of marketable power--information on survival rates is needed in order to assess the cost-effectiveness of mitigation alternatives.

The evaluation of downstream passage mitigation measures becomes a matter of characterizing how each specific measure, alone and in combination with others, affects the number of surviving smolts. For example, one may want to know how introducing another million smolts from an upstream hatchery may increase the number of smolts entering the ocean; how increasing the size and frequency of spills at a particular dam changes mortality attributable to that dam; or how increasing minimum flows through a stretch of river during a time period increases survival through that stretch.

In the Columbia River Basin, these questions are difficult to answer. It is hard to separate the effects of a particular mitigation policy from the impacts of other policies. In addition, as mentioned in the opening paragraphs, estimating the survival rate past a dam or through a river reach, or even estimating the total number of fish passing a particular point in the river system, are tricky problems. The next section explains the difficulties involved.

TECHNIQUES FOR ESTIMATING SMOLT ABUNDANCE AND SURVIVAL

Unlike adult fish passing upstream at a dam, smolts passing downstream are very difficult to enumerate. If the dam is sufficiently high to require a fish ladder, visual or automated counters can be installed in the ladders, and adult fish can be forced to pass the counting site in concentrations suitable for counting. However, it would be impossible or extraordinarily expensive to make a visual count of all juvenile fish passing a given point in the Columbia River system, except perhaps in the smallest streams. Too many small fish, too much water, and probably too many escape routes exist.

If juvenile fish were as likely to be in one part of the river as in any other, one could construct a sampling device to count the fish even with the limited current technology. The number of fish within some restricted area of the stream's cross section that pass a marker (e.g., a bridge) times the ratio of the total stream cross-sectional area to the restricted area area would equal the total number of fish passing that marker. However, fish seldom move in uniform densities in a river, and our knowledge of the spatial and temporal distribution of migrating smolts within the river is insufficient to provide a basis for an alternative hypothesis. Therefore, an approach based on simple cross-sectional sampling probably would not estimate the true population accurately.

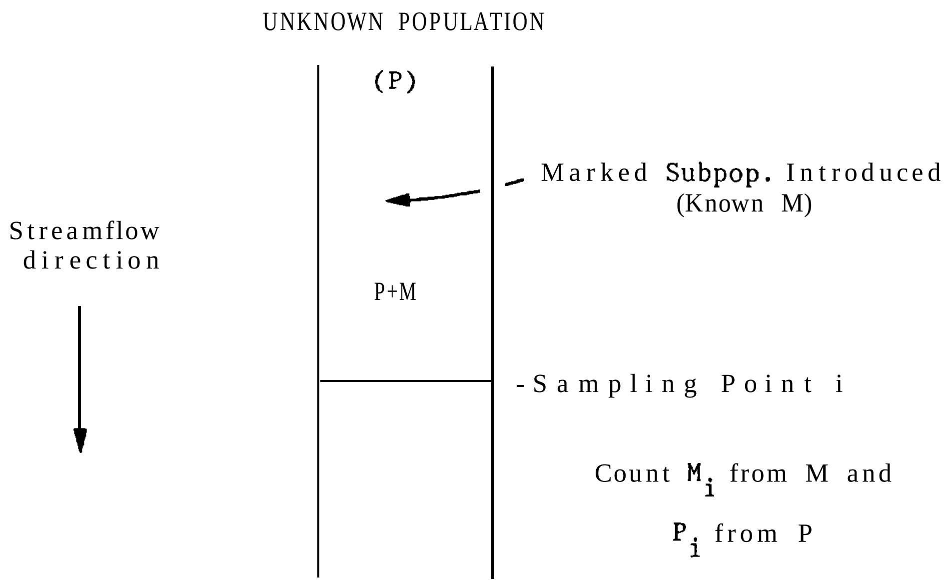
The alternative is to design a sampling program which relies on direct or indirect sampling. Direct sampling methods involve introducing one group of marked fish upstream and recovering the fish at a downstream site, such as at a dam. The approach relies on the estimation of the absolute sample collection efficiency (the proportion of marked fish passing a point that are sampled) at the downstream site. Indirect sampling, in contrast, involves introducing a target group of fish upstream and a control group downstream (just above the recovery point), and recovering both groups of fish at the sampling point. This approach defines the relative sample collection efficiency (the proportion of the control group that is sampled), since the sample efficiency is allowed to vary from experiment to experiment.

The simplest direct-sampling method is to create a subpopulation of known size and to follow the movement of this subpopulation. An investigator would mark the known subpopulation, release it, sample the total unknown population (including the known subpopulation) as it passes the point of interest downstream, and use the sampling efficiency obtained for the subpopulation to estimate total population size. Schematically, this process is represented in Figure 8.1.

The sampling method must produce equal fractions of the marked and unmarked populations in order to estimate accurately the total population; that is, marked fish need to be homogeneously distributed among the total migrant population passing the sampling site during the sampling period such that $M/M_i = P/P_i$. Any inequality in this ratio will bias the estimate of total population size. Such bias might result if the marked fish swim together on one side of the river and the unmarked fish swim uniformly throughout the river. In this case, a sample collected from a small portion of the total cross-section of the river will not exhibit an accurate ratio of marked to unmarked fish. Similar effects may result if the marked fish swim at a different depth, if the sample method involves nets (which are generally size-selective) and the sizes of marked individuals systematically differ from those of unmarked individuals, or if the two populations prefer different routes past a dam and not every route is sampled equally.

Putting aside the difficulties for a moment, how does this method permit investigators to make inferences about survival and hence potentially about improvements in survival resulting from mitigation efforts? If one wants to estimate how a particular obstacle affects survival, one approach would be to estimate the population size (P) above and below the obstacle.

$$\text{estimated survival} = \frac{P\ (\text{below})}{P\ (\text{above})}$$



$$\text{Estimate P as } \frac{P_i(M)}{(M_i)} \qquad (8\text{-}1)$$

M	=	number in known marked subpopulation
M_i	=	number sampled of marked population at point i
P	=	number in unknown total population
P_i	=	number sampled of total population at point i

Figure 8.1 Schematic of Sampling Site for One Test Subpopulation (Direct Sample)

or

$$S_j = \frac{\hat{P}(b_j)}{P(a_j)} \qquad (8\text{-}2)$$

where S_j = fraction of fish surviving passage by obstacle j

$\hat{P}(b_j)$ = estimated population below point j

$\hat{P}(a_j)$ = estimated population above point j

This procedure would be complicated by the sampling done on the subpopulation M above the dam and its implication for the below dam count. If K fish were sampled from M and not replaced, the subpopulation would become M-K. If the K fish were replaced, questions about their relative physical condition--their survivability--would arise.

A common approach in such a direct sampling scheme is to adjust the population estimates based on the differences between the observed recovery rate of the marked subpopulation and the expected recovery rate. The expected recovery rate is a function of the sample site, river flow, spill, and turbine diversions. It is an estimate of the absolute collection efficiency of a sample; that is, a yardstick to use in subsequent samples for mortality. Ideally, it is computed from a series of experiments at a sampling site, with varying flow conditions, where one releases marked "control" fish just above a sampling point to see how many of these can be caught at the sampling point. If one can establish a predictable relationship between river flow and recovery rate (a set of flow-recovery curves) and if this remains constant over time, only one group of test fish will be needed to estimate survival rates in subsequent mortality studies. The observed survival rate of the test fish would be adjusted to account for the sample efficiency based on the difference between the actual rate and the expected rate.

For example, if the expected rate of recovery of fish (the absolute sample efficiency) based on prior experiments is thirty percent (at a given flow) and if fifteen percent of a fish subpopulation marked and released upstream is actually recovered with that flow, then the estimated mortality rate is fifty percent (1.0 - 0.15/0.30). At a different flow with a different absolute sample efficiency of twenty percent, the same actual fifteen percent recovery may indicate a mortality rate of twenty-five percent (1.0 - 0.15/0.20). Of course, the actual mechanics of these calculations is considerably more complex than simply dividing these percentages, but the basic approach is the same.

The difficulties involved in meeting the assumptions necessary to estimate population size by this approach make it a poor choice for use in a large system such as the Columbia. A major problem is the possibility of additions to the population of migrating fish. If one is examining a long river reach or a series of dams and reservoirs, the estimation procedure will break down if tributaries or in-river nursery areas add large, unaccounted-for migrant populations to the river.

One way to get around this difficulty is to use an indirect sampling method. This technique defines the relative sample efficiency and involves the addition of two marked groups at different locations. One of these groups makes the trip through the river reach and is assumed to be subject to all the same causes of mortality as is the general population. The second marked group enters the river just above the sampling station and is used to determine the probability of capture at that station (see Figure 8.2). The probability defines the relative efficiency of sampling, and does not rely on an estimate of a long-term absolute efficiency. It is assumed that there are no losses to mortality for the members of the second group. Since by definition no recruitment can occur to either marked group, the algebra becomes :

$$M_{i,T} = \bar{M}_{i-1,T}(\hat{S}_{i-1,i}) \tag{8-3}$$

and

$$M_{i,T} = m_{i,T}\left[\frac{\bar{M}_{i,c}}{m_{i,c}}\right] \tag{8-4}$$

therefore,

$$\bar{M}_{i-1,T}(\hat{S}_{i-1,i}) = m_{i,T}\left[\frac{\bar{M}_{i,c}}{m_{i,c}}\right] \tag{8-5}$$

Since $$\hat{S}_{i-1,i} = \frac{\hat{M}_{i,T}}{\bar{M}_{i-1,T}} \tag{8-6}$$

then

$$S_{i-1,i} = \frac{m_{i,T}\,\dfrac{\bar{M}_{i,c}}{m_{i,c}}}{\bar{M}_{i-1,T}} = \frac{\dfrac{m_{i,T}}{m_{i,c}}}{\dfrac{\bar{M}_{i-1,T}}{\bar{M}_{i,c}}} = \frac{\dfrac{m_{i,T}}{\bar{M}_{i-1,T}}}{\dfrac{m_{i,c}}{\bar{M}_{i,c}}} \tag{8-7}$$

where

$\bar{M}_{i-1,T}$ is the known marked test subpopulation inserted at the upstream end of the reach at point i-l

$\hat{M}_{i,T}$ is the estimate of the marked test population at the sampling location at point i

$\hat{S}_{i-1,i}$ is the estimate of the fraction surviving through the reach from point i-l to point i

$\bar{M}_{i,c}$ is the second, or control marked subpopulation inserted just upstream of point i

$m_{i,T}$ and $m_{i,c}$ are sample sizes from the test and control populations at point i

The method does not depend on an estimate of the absolute efficiency of a sampling scheme and site, but it does depend on the validity of the assumption that sampling from $M_{i,c}$ occurs with the same efficiency as sampling from $\bar{M}_{i,T}$; that is, the'sample captures the same proportion of fish in the control population as it does in the surviving test population. The indirect method has the disadvantage of requiring control and test groups for each mortality study, which means that such studies potentially can impact more fish.

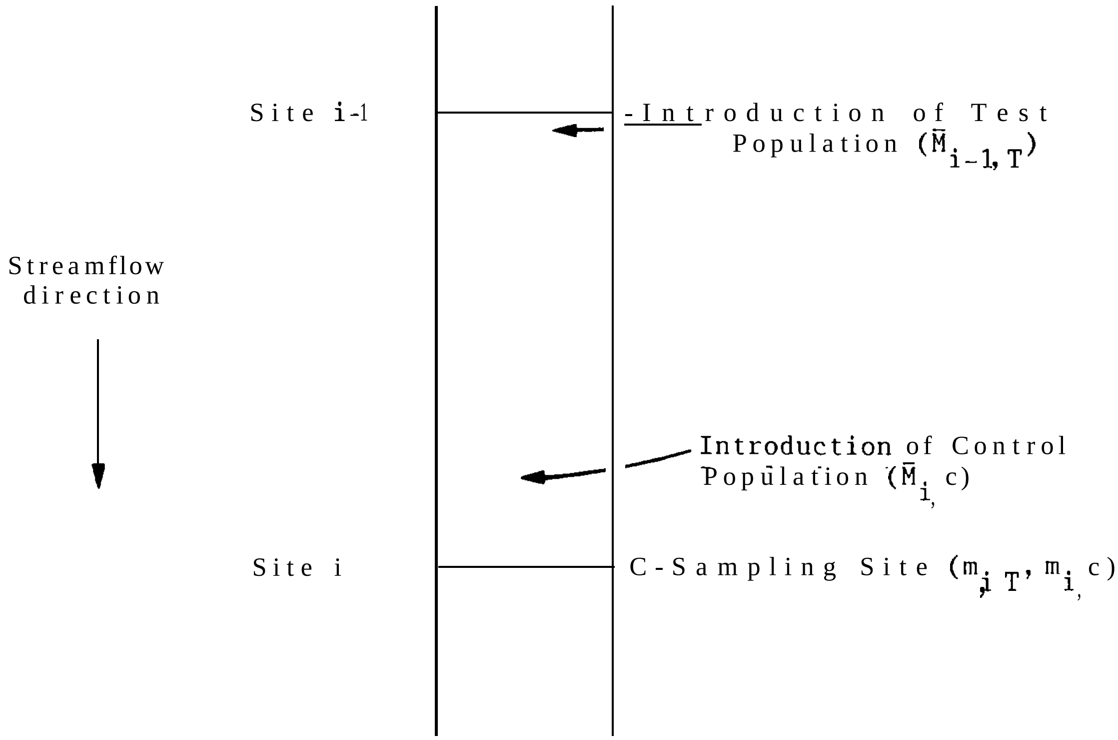


Figure 8.2 Schematic of Sampling Site for Two Test Subpopulations (Indirect Sample).

SOME ISSUES BEARING ON SURVIVAL ESTIMATION IN THE COLUMBIA BASIN

Application of the general techniques described above has given rise to a number of issues that have been discussed in the literature. These conveniently may be grouped under three headings: biology, timing, and variance or confidence estimation, even though the problems overlap these categories to some extent.

Biology

The major biological issues revolve around the relative condition of hatchery and wild stocks, and the effect of marking, counting, and other handling on the physical condition of the fish. If hatchery fish have lower dam or river reach (i.e., in the river) survival rates, then survival estimates based on marked subpopulations of hatchery fish will, other things being equal, underestimate wild fish survival. Furthermore, in looking at river reach survival rates, if hatchery fish behave peculiarly immediately after being released into the river, then it will be problematic to pick a release point at which to insert the control subpopulation: if too far from the sample point, mortality of the control group between the release and sample points may become significant; if too near, sampling efficiencies for the two subpopulations will be different.

A related question is whether hatchery fish may have conditions leading to delayed mortality. For example, fish from hatcheries might encounter conditions in the hatchery which weaken them, and begin to die when they reach the river and start to migrate. The test fish might spend enough time in the river to begin suffering deaths, but the control fish might not. The resulting survival estimate would reflect migration hazards and hatchery conditions and thereby overestimate mortality rates associated with the migration. A conceptually similar issue, discussed below under timing, concerns the onset of the changes that lead to migration (smoltification).

Most of the studies done to date in the Columbia have involved marking by some combination of fin clipping and freeze branding (some coded wire tagging has been done, but not in the context of smolt survival or travel time). These require that fish be handled both for marking and mark reading. The marking process itself introduces stress that other hatchery fish do not face which can contribute to mortality. Thus, marking itself can be the source of bias in the estimates of survival.

As for the necessity of handling the fish for mark reading, this can be an issue in two senses. First, if large numbers of fish have to be sampled and thus handled to get statistically useful numbers of fish from the two marked subpopulations (in an indirect sampling scheme), then a significant fraction of the total population will be subject to stress. Second, in some indirect sampling methods used to estimate dam mortality, a subset of the fish sampled are marked at the sampling point and re-released immediately upstream of the dam to constitute the control group. This means that some hatchery and some wild fish will be handled, marked, and released--stressed--after they have been actively migrating for several days or weeks. This may produce a differential level of stress compared to marking hatchery fish before they are released into the river.

Timing

Two principal timing issues are worth mentioning. The first involves the spacing in time of experiments. The second concerns the relative timing of the releases of test and control subpopulations for any single experiment.

The justification for the direct sampling approach in the Columbia is that sufficient experimentation would allow the estimation of empirical functions relating sampling efficiencies to such measurable parameters as flow and percent of flow being spilled (see, for example, Raymond 1979; Sims et al. 1982; Committee on Fishery Operations 1983; McKenzie et al. 1984a; McKenzie et al. 1984b; and McKenzie et al. 1985) Such an approach would allow one to use one group of fish in a mortality study rather than two. Unfortunately, past efforts to develop these functions in the Columbia have been disappointing and one still cannot predict the absolute sampling efficiency of a given site with any reasonable level of certainty.

Because of this limitation, researchers in the Columbia Basin generally rely on indirect sampling methods, which require the use of test and control groups. Yet, this approach encounters two difficulties directly related to the two timing issues mentioned above. First, if the test and control group sampling experiments are repeated over the same reach in the same year, and if it is desired to separate the test releases by at least a few days (see below discussion), the use of several of these "replicates" could conceivably run the experiment over the period when smolting fish can usefully be released. They might pass their "prime condition" and could lose their drive to migrate to sea. This situation, known as "residualism", would bias estimates of survival by removing non-migrating fish from the assumed control population $\bar{M}_{i,c}$. The relative sampling efficiency estimated from the control group, $m_{i,c}/\bar{M}_{i,c}$, would be lower than the sampling efficiency of the test group, and thus the survival estimate S would be biased upward. This can be seen from Equation 8-7.

The second complication is that the indirect method requires that sampling of test and control populations proceed at the same efficiency; that is, the same proportion of test and control populations must be sampled at a point. However, as noted above, sampling efficiency depends on such variables as total flow, percent of flow being spilled as opposed to passing through the turbines, and probably on temperature, turbidity, cloud cover and others as well. These background conditions are constantly changing at every sampling point.

To complicate matters, control groups may arrive at different times than the fish in the test group. If the control group is removed from the hatchery and released at a downstream point at the same time that the test group leaves the hatchery, the control group will arrive at the sampling point before the test group. This is potentially significant if the distance between the release point and the hatchery is more than 20 or 30 kilometers. Additionally, if the distance is large, the test group will arrive at the sampling point over the course of several days, while the control group will arrive in a more-compact body. Thus, the control and test groups may encounter different flow conditions and sampling efficiencies may differ. This violates the assumption that control and test group sampling efficiencies are the same.

On the other hand, if the control fish are marked and kept in the hatchery and held back from release until the test group makes its way from the hatchery to the release point, the control group and test group may arrive at the sampling point at the same time, but with the added complication of residualism and perhaps not in the same biological condition. The control group will have encountered hatchery conditions for a longer time than the test group (as discussed above). Sampling efficiencies and mortality rates may differ.

In order to help overcome the problem created by the spread-out arrival times of the test group, the control fish generally are released over the course of several days. For example, a common design seems to be to release three control subgroups two days apart. The timing of the control group releases is intended to be such that their arrival coincides with and spans the time of arrival of the test fish. Inevitably, some averaging over time, and hence over changing conditions, is necessary. This is because each control group must be associated with the entire test group, not with any subset of it. Therefore, survival will be calculated on the average results for the three control groups. This can be calculated as the average of separately calculated survivals, weighted by the size of each control subgroup,

$$S = \frac{1}{(\bar{M}_{1,c} + \bar{M}_{2,c} + \bar{M}_{3,c})}\left[(\bar{M}_{1,c})(S_1) + (\bar{M}_{2,c})(S_2) + (\bar{M}_{3,c})(S_3)\right] \qquad (8\text{-}8)$$

where

$$S_i = \frac{\dfrac{m_T}{\bar{M}_T}}{\dfrac{m_{i,c}}{\bar{M}_{i,c}}} \qquad \text{for } i = 1, 2, 3 \qquad (8\text{-}9)$$

or as the survival implied by the average of the sampling efficiencies, weighted by the size of each control subgroup,

$$S = \frac{\frac{m_T}{\bar{M}_T}}{ASE} \tag{8-10}$$

where

$$ASE = \frac{1}{\bar{M}_{1,c} + \bar{M}_{2,c} + \bar{M}_{3,c}} \left(m_{1,c} + m_{2,c} + m_{3,c} \right) \tag{8-11}$$

and $m_{i,c}$ = number sampled of control subgroup i = 1, 2, 3

$\bar{M}_{i,c}$ = total number in control subgroup i = 1, 2, 3

m_T = number sampled in test group

$\bar{M}_T$ = total number in test group

ASE = average sample efficiency

S_i = survival based on control subgroup i = 1, 2, 3

S = average survival

Such averaging imparts an unknown bias (both in size and direction) to the results, especially if the relationship between sampling efficiency and underlying environmental conditions is non-linear. These two estimates of survival are not in general the same, as some algebra will convince the reader.

This approach still leaves the problem of residualism. The conflict between ensuring the simultaneous arrival of test and control groups on the one hand, and trying to eliminate residualism by simultaneous releasing test and control groups on the other, clearly make the choice of sampling schemes difficult. However, recent survival studies (Fish Passage Center 1986a and 1987) have indicated that the stress due to holding of marked fish in control groups is more detrimental than the failure of test and control groups to arrive simultaneously at the sampling point. Thus, in general, test and control group releases have been made simultaneously, though separated spatially as already explained.

The simultaneous-release, indirect method as currently practiced contains a version of earlier attempts to relate sampling efficiency to flows or spills. Researchers are increasing the number of sampled fish in order to derive indices to relate these factors. Both the fraction of time (over a 24-hour period) the sampling station operates and the fraction of river flow passing through the power house are used to derive these indices. This latter factor represents a simple version of the relationship between sampling efficiency and spill (the fraction spilled being the complement of the fraction passing through the turbines). If test and control groups are released on the same day, they will arrive on different days, but the indices (when summed over all the days of the run) presumably can correct for the different spills and provide comparable sampling efficiencies.

This indexing approach seems to negate the primary advantage of the indirect method--the independence of it from estimates of sampling efficiency. What remains looks a lot like the direct method, with control groups and indices providing a kind of absolute sampling efficiency estimate.

Estimating Variance and Confidence Limits for Survival Fractions

Estimates of populations and survival fractions made by the methods discussed above are clearly subject to many kinds of errors. Those introduced by biological problems and the related matter of timing were mentioned above. However, if one ignores these complications and assumes that every fish has the same probability of surviving and every fish the same probability of being sampled, repeated estimates of the survival probabilities based on the sampling regime will be distributed around the true value.

In such circumstances there are two ways to obtain estimates of other descriptive statistics, such as the variance of the estimates of survival.

The first is based on the assumed underlying probability model. This approach receives a fair amount of attention in the publications of the Columbia smolt monitoring program, though the underlying assumptions and derivations are nowhere clearly set out. Perhaps the most complete discussion is found in McKenzie et al. (1985), where the authors concentrate on statistical issues. A related but more elaborate model for estimating the effectiveness of spills in moving smolts past dams is set out with some care in Stuehrenberg et al. (1986). In the notation developed above, the formula presented by McKenzie et al. for the variance of the survival estimate is:

$$\mathrm{Var}(S) = \frac{1}{(\bar{M}_{i-1,T})^2} \mathrm{Var} \left(\frac{m_{i,t}}{\frac{m_{i,c}}{\bar{M}_{i,c}}} \right) \tag{8-12}$$

$$= \frac{1}{(\bar{M}_{i-1,T})^2} \left(\frac{m_{i,t}}{\frac{m_{i,c}}{\bar{M}_{i,c}}} \right)^2 \left[\frac{\mathrm{Var}(m_{i,T})}{(m_{i,T})^2} + \frac{\mathrm{Var}\left(\frac{m_{i,c}}{\bar{M}_{i,c}}\right)}{\left(\frac{m_{i,c}}{\bar{M}_{i,c}}\right)^2} \right]$$

Using this formula, the standard deviations for the smolt experiments run over the past 5 to 6 years are impressively small; therefore, the 95 percent confidence intervals are impressively narrow (for example, see McConnaha and Basham 1985).

However, the standard errors are considerably larger when calculated using the actual survival values for the 2 or 3 replicate experiments for a given river reach and year. Most important, because of the small sample sizes (number of replicates) and the accompanying large t statistics, the resulting large confidence intervals render the point estimates of survival probabilities essentially meaningless. To get a sense of the importance of the number of replicates in this calculation, consider the results presented in Table 8.1 (reproduced from Fish Passage Center 1986a).

Junge (internal Fish Passage Center memo reproduced in Fish Passage Center 1986b) forcefully describes the statistical problems of survival estimation and hence of river management policy analysis:

> <u>Variances of estimates considered here should never be made based on idealized probability models</u>. Except for some tests of turbines and spillways where immediate mixing of control and experimental releases occur, such estimates are illogical and can be very misleading. The many sources of variation, other than those estimated with idealized probability models, should make this very apparent. For example, in testing mortalities passing the five mid-Columbia dams, it is theoretically possible that a particular release would pass all dams over spillways with a mortality less than <u>10</u> percent (at 3 percent per spill); or a release could pass <u>all</u> dams through turbines with a mortality of over 44 percent (at 11 percent per turbine). In low flow years, reservoir mortalities can greatly exceed mortalities at the dams; and even in normal flow years, the variations in mortality depending upon the migratory path can be excessive (for example, due to predation or residualism). It is not difficult to find cases where confidence intervals based on models is a tiny fraction of the confidence interval indicated by replicate estimates.
>
> Even more important, researchers planning such studies always approach statisticians to estimate the number of recoveries needed for an adequately precise estimate. It may be reasonable to require a minimum number of recoveries, but it should not be inferred in any way that the idealized variance based on these recoveries is applicable. As stated earlier, estimates of variance should be made directly from replicate experiments. The more proper question for the researchers is: How many replicates do we need? Of course, we first need some valid estimates of variance based on properly controlled replicates.

The problem with this advice is that increasing the number of replicates (using the freeze-branding/clipping mark technology) increases the number of fish that have to be handled. This results in objections on the basis of both expense and, probably more important, of risk to a large fraction of the juvenile run. Thus, even where apparently straightforward survival probability estimation is concerned, the smolt monitoring program managers find themselves in a difficult position. A single estimate of survival probability for a long stretch of the Columbia or for one of its tributaries does not provide much understanding of the relation between survival and policy choices, yet more extensive studies may endanger the smolt.

What hope is there for future success in pursuing the goal of improved smolt monitoring? One possible answer to this question, better technology, is discussed in the next section.

Table 8.1 Effect of Increasing Number of Replicates on Precision of Mean Survival Estimate of Winthrop Hatchery Spring Chinook, Assuming Mean Survival (S) and Standard Deviation (b) Estimates from 1985 and 1986

Population $\hat{s}$	$\hat{\sigma}$	Number of Replicates (n)	Standard Error ($\hat{\sigma}/\sqrt{n}$)	t Distribution value[a]	95% Confidence Interval[b] Range	% of mean $\bar{s}$
1985						
.45	.088	2	.062	12.706	.79	175
.45	.088	3	.051	4.303	.22	49
.45	.088	4	.044	3.182	.14	31
.45	.088	5	.039	2.776	.11	24
.45	.088	6	.036	2.571	.09	21
1986						
.46	.111	2	.078	12.706	.99	215
.46	.111	3	.064	4.303	.26	60
.46	.111	4	.055	3.182	.18	38
.46	.111	5	.050	2.776	.16	30
.46	.111	6	.045	2.571	.12	25

[a] Significant level = 0.05 and degrees of freedom n-l

[b] Equals $t_{c/2}$(n-l d.f.) $\hat{\sigma}/\sqrt{n}$

IMPROVEMENTS IN MONITORING TECHNOLOGY

Advances are being made in smolt monitoring technology along two lines. First, improvements in monitoring equipment, the actual devices which allow one to detect the passage of fish, add to the information content of data which are being collected. Second, recent developments in statistical methodologies permit one to draw stronger inferences from the data as they become available.

Neither of these two advances constitutes a solution to the problems of smolt monitoring, but both contribute to an improvement in monitoring capabilities. Each in part depends on the other. More precise and accurate monitoring equipment is useless if statistical methodologies are too weak to draw useful inferences from the improved data. Similarly, much of the promise of new statistical methods rests on the availability of more accurate data that will result from the application of new monitoring devices.

To set the stage for a brief consideration of three major advances in monitoring equipment--hydroacoustic monitoring, radio transmitter tags, and passive integrated transponder (PIT) tags--consider the characteristics that an ideal technology would display:

- it should involve identifying labels (marks or other devices) which are easy to implant in each fish (or better still, not involve marking at all)
- the implantation of identifying labels should not cause the death of any marked fish, nor impair any fish's functioning (As a corollary, it should not be necessary to select extra large, strong fish to achieve this. The requirement should be met when average sized juveniles are used.)
- the identifying labels should carry enough information so that individual fish can be uniquely identified as they pass a monitoring station
- no handling of the fish should be required (reading the identifying label of each fish should be done remotely)
- reading should be accomplished without having to channel fish into special passages or chambers--a river census of marked fish should be possible

- detection efficiency (fraction of marked fish passing a point that have their identifying labels read) should be close to 1.00

- the identifying label should survive in or on the fish through the entire outmigration, the period at sea, and the adult spawning migration

- the identifying labels should be inexpensive

- the detection system should interfer only minimally with project operations

Measured against these enormously high standards, the advances described below leave much to be desired. It is possible, however, that further research could push them towards these ideals.

Hydroacoustic Monitoring

Hydroacoustic monitoring involves the use of sonar, installed in dam forebays, to estimate the number of fish passing through one or another part of the structure. (see, for example, Raemhild et al. 1984, 1985a, and 1985b). The major advantage of the method is that it allows detection of fish at the normal passage points--especially turbines and spillways--and does not require special channeling of limited numbers of fish out of the main body of migrants. It also does not require handling or marking of any fish.

In principle, each project in the system could be equipped with hydroacoustic equipment to monitor fish passage at every aperture, 24 hours a day. In practice, at least so far, only a few dams, only a fraction of the total area of routes through each covered dam, and only a fraction of each 24 hours are covered. Therefore, the use of hydroacoustic monitoring to date has been confined to estimating the relative numbers of fish approaching the dams which pass through spillways and turbines under different flow and spill conditions. The principal limitations of hydroacoustic monitoring are that:

(1) only estimates of the total number of fish are available, since it does not allow the identification of any particular species, population, or sample; and

(2) its detection efficiency and error structure are poorly defined because there has been no way of verifying the hydroacoustic counts by independently counting the fish moving through the monitored apertures.

Radio Transmitter Tags

The radio transmitter tag technology involves the implantation of small radio transmitters into a smolt's stomach, through insertion down the throats of marked individuals (Stuehrenberg et al. 1986). The transmitters broadcast intermittent coded messages, and there is enough information carrying capacity in the broadcast pulses to allow identification of individual fish.

This method currently has several serious limitations. First, the large size of the transmitter and battery package necessitates the selection of large smolts for marking, which introduces possible bias. (Stuehrenberg et al. (1986), however, assert that the population of marked fish are not significantly larger than simultaneously netted samples of unmarked fish from the river.) Furthermore, even with a large-fish selection policy, there is evidence of impairment of swimming ability and buoyancy control.

Second, the current transmitters have too-limited lifetimes and ranges. The minimum lifetime of normally functioning equipment is said to be 3 days, and based on a Stuehrenberg et al. graph (1986, p. 19) it appears that median time to failure is 5 days, with no tags lasting longer than about 8 days. This precludes experiments involving any substantial portion of the basin, where several weeks of monitoring would be required just to insure full coverage of a migratory event.

A final limitation of the technology relates to the concentration of tagged fish. The detection equipment has been unable to sort out individual fish when the number of tags transmitting on the same frequency (of nine available frequencies) in the same area exceeds six.

The existing use of radio transmitter tags has been limited to applications involving a single dam. The range of the transmitters, using the original detection equipment, has been reported to be between 100 and 1000 meters, depending on the depth of the fish. (If fish were swimming at a depth greater than 7.5 meters, their signals were undetectable by above-water antennae.) There are plans to add underwater antennae in future work, but even 1000 meters is not a great distance on a river the size of the Columbia.

For whatever combination of range, depth, and message-sorting reasons, the detection rates reported in Stuehrenberg et al. (1986) are in the neighborhood of 70 percent, with substantial variation between experiments. If this problem of low-detection and high-variability persists, the confidence limits on numbers detected and hence on survival probably would be large enough to question the utility of the results. The 95 percent confidence intervals reported by Stuehrenberg et al. (1986, pp. 8-11) were calculated on the basis of their probability model. The estimates of survival for the two test conditions show discrepancies between the model's predictions and the actual results, recalling attention to the strictures set out by Junge and discussed in the previous section.

As currently designed and built, radio transmitting tags may be useful for evaluating mortality at individual dams, but not for system-wide studies. Hope for the future rests on the production of a transmitter tag that is smaller, able to be inserted so that it does not interfere with the fishes' capabilities, of greater transmitting power, capable of transmitting more information per pulse, and longer-lasting. These requirements play to the strength of current developments in electronics, so substantial improvements may take place.

Passive Integrated Transponder Tags

PIT tags are analogous to the equipment installed on aircraft for air traffic control purposes. They do not actively transmit information but remain passive (quiet) until triggered by a burst of energy received from the detection equipment transmitter. The resulting pulse available to the detection equipment is said to be capable of carrying enough information to allow 35 billion individual signals. This certainly should be enough to cover any conceivable experiments in the Columbia (see Prentice and Park 1984; Prentice et al. 1985).

This tag theoretically allows researchers to follow individual fish from the top to the bottom of the river system. Furthermore, the use of a number of replicates per season would allow for variation in control conditions against a background of natural variation, but with high intraseason autocorrelation. The tag volume itself is only 2 percent that of the radio transmitter tag and it may be inserted into the body cavity of the fish by injection. It is too small to have any effect on the fish's performance, if correctly inserted, and its lifetime is apparently unlimited. (The current insertion method aims at having tags survive in the fish from smolt to returning adult stage.) Its detection efficiency averaged 90 percent, in a recent project, although the test conditions in this project were established artificially and did not mimic the situation at a dam. (Prentice et al. 1985)

As with the other technologies, PIT tags have limitations. Researchers will need to develop a method of insertion that is fast, easy, and does not result in high fatalities from puncture of vital organs. More important, detection will require guiding the fish into some sort of tunnel, because the effective detection range is a matter of centimeters. In the current situation at the Columbia River System dams, this limitation reintroduces all the problems of sampling at each dam that the two previous methods avoid. This is because the majority of the run-of-river dams in the Columbia and Snake Rivers do not have the sophisticated bypass systems in place that are necessary to establish permanent detection stations. While no handling of sampled fish would be required, the size of the tagged population would have to be chosen so that at (probably very low) sampling efficiencies, numbers of detected fish would be sufficient to produce significant statistics. In addition, because sampling efficiency would be unknown a priori, test and control groups of tagged fish would have to be provided as they are now with freeze branding, in order to establish an estimated relative sampling efficiency.

Thus, PIT tags currently seem to offer only the advantages of long-life and of not requiring that fish be handled for reading. The experimental designs feasible with this technology are restricted by the necessity to introduce control populations to estimate sampling efficiencies. However, if the current goal of installing fish bypasses and fish guidance equipment at every dam is met, and if those structures have the necessary PIT tag reading equipment, the technology may offer a significant improvement in smolt monitoring.

Advances in Statistical Methodologies

Congruent to the development of monitoring equipment, statistical analysts continually have been looking for better ways to make inferences from the available data. Foremost , a recent monograph by Burnham et al. (1987) represents a quantum jump in statistical rigor for analysts trying to estimate the survival of migrating smolts using release-recapture methods. Due to the potential importance of this text and the methods that it describes, a brief overview of the release-recapture methods is presented below.

A second advance in analytical capability is the stochastic compartment model of reservoir passage, which is discussed in Chapter 4. In the reservoir passage model presented there, smolt migration through a reservoir occurs as an irreversible particle diffusion process. This approach incorporates techniques which are novel to the study of downstream migration, but which have been applied successfully in other disciplines.

These two efforts have aimed in different directions and been developed for different purposes. Therefore, they exhibit distinct traits. However, the two methods complement each other quite well, as demonstrated in a following section.

Release-Recapture Methods

The text by Burnham et al., Design and Analysis Methods for Fish Survival Experiments Based on Release-Recapture (American Fisheries Society 1987), presents the most complete and comprehensive treatment of this topic which has been published to date. The text was made possible by a grant from the Chelan County Public Utility District, so the authors developed it with particular reference to the problems facing fishery managers in the Columbia River Basin. The book focuses on how to determine the extent and nature of the impacts of specific treatments on survival rates of migrating fish. Within the text, treatments usually refer to in-river dam passage in one mode or another, but one also could apply the available methods to situations in which treatments refer to pre-release events.

A premise of the monograph is that two or more groups of fish will be divided among test ("treatment") and control populations, both of which are marked, released into the river, and recaptured at one or more sampling sites downstream. The "treatment effect" is defined as the ratio of test fish survival to control fish survival. In an example given in the text, test fish are released in the forebay of a particular dam and control fish are released immediately downstream of the dam in the tailrace. One or more dams which occur further downstream are used as sampling or recapture sites, where the ratio of control to treatment fish is obtained.

The authors present four protocols for the design and analysis of such experiments. The differences among protocols rest on whether each fish carries a unique mark and whether fish are subject to recapture more than once (i.e., if a marked fish captured at Dam 2 would be released again and subject to capture at Dams 3, 4,..., k). Protocols also are distinguished by their ability to test model assumptions and hypotheses. Each protocol has clearly defined assumptions, as well as an explicit theoretical development of the estimatable parameters and their variances and covariances. Selection of a particular protocol represents a tradeoff between effort, expense, and the extent of statistical inferences which can be drawn from the experiment.

The most informative of the protocols is the "complete capture history protocol" in which each fish bears a unique mark and may be recaptured more than once. The authors consider this protocol to be superior- to the others because it requires fewer assumptions, can test more key assumptions, and offers greater flexibility in the estimation of parameters. Application of this protocol will be feasible using new advances in equipment such as PIT tags.

Some of the more applied aspects of the Burnham et al. text come from the chapters on replication and planning of experiments. The discussion of replication stresses the point that actual sample variances of survival estimates are likely to exceed theoretical variances by a considerable amount (see above discussion). To get around this, experiments can be replicated by subdividing the release groups. This is easy to do if each fish carries a unique mark and there are ways of randomly assigning individuals to replicate groups (e.g., by using the last digit of the individual tag numbers).

The chapter on planning experiments covers such topics as the selection of an experimental protocol and discusses effort and sample size considerations. Like the replication chapter, it also stresses replication of experiments. One interesting observation from this section is that in the effort to increase the accuracy of survival estimates, it is more efficient to increase the recovery rate than to increase the number of marked and released fish. In other words, one can get equivalent information from a sample size of less than half the normal size by doubling the rate of recovery. These observations play to the strengths of PIT tags.

A final note about the text is that an interactive computer program (RELEASE) has been developed which makes the analysis of release-recapture data fairly straightforward. A copy of this program is available free of charge from Dr. Gary C. White of Colorado State University. As with any statistical package, the results of the analyses can only be as good as the data which go into them. It is the responsibility of the fishery investigators to insure that the release-recapture experiments are carried out in a manner which is consistent with the assumptions inherent in the analyses.

Compatibility of Statistical Models

The statistical approach of Burnham et al. and the stochastic compartment model of reservoir passage proposed in Chapter 4 have been developed for different purposes and have different characteristics. However, because they address separate aspects of the general problem of downstream survival, the two methods complement each other nicely.

The release-recapture protocols primarily have been developed to address situations in which the test fish suffer some acute or immediate damage, as opposed to a chronic effect. For example, fish which are mortally injured as they pass through a turbine may die immediately or within a few kilometers of the dam. The probability associated with being killed by turbine passage is assumed to be the same for all fish passing through the turbine. In contrast, the reservoir passage model is concerned with the chronic, lower-level mortality risks (e.g., from predation or disease) which are always present as the fish migrate through the reservoir. For example, in the reservoir passage model, the longer that a fish stays within a given reservoir, the more likely it is that that fish will die within that reservoir. Thus, a fish which migrates through the reservoir quickly has less chance of dying than a fish which migrates slowly.

In a sense, the reservoir passage model's premise that the probability of passage is a function of the time in transit violates an assumption of the release-recapture protocols. Since the reservoir passage model explicitly models time in transit as a random variable which follows a gamma distribution, the probability of survival is a random variable as well. In contrast, the release-recapture protocols assume that fish within each group (test or control) are homogenous with respect to survival probabilities; that is, the probability of survival is the same for both groups.

The practical implications of this inconsistency may be relatively insignificant. Burnham et al. claim that the estimators which are used within the release-recapture protocols are fairly robust to (not affected by) violations of the assumption of homogeneity. A useful experiment to test this claim would be to set up a simulation experiment which mimics the type of release-recapture data that would be expected under reservoir passage conditions predicted by the reservoir passage model and to analyze this data using the release-recapture protocols. The RELEASE computer program should facilitate such an experiment. Options within the program allow one to use Monte Carlo simulation to test the sensitivity of survival parameter estimates to heterogeneity in survival and recapture probabilities.

The fact that one can apply both the reservoir passage model and the release-recapture protocols to the same data sets without any apparent inconsistencies in assumptions (other than the one mentioned above) is encouraging. Both methods use release-recapture data, although the reservoir passage model requires additional information on the time delay between release and recapture. Both methods would benefit from additional information on individual fish.

Future Directions for the Smolt Monitoring Program

To review, the two principle objectives of smolt monitoring are: (1) to keep track of the outmigration as it progresses, in order to manage spills and flow enhancement to benefit the smolt; and (2) to permit an assessment of the biological effectiveness of alternative mitigation strategies. Whether or not the current smolt monitoring program adequately meets the first objective is open to debate. A recent study by Skalski (1988) presents rather convincing arguments that it does not. There is also little doubt that the current monitoring program falls far short of fulfilling the second objective. Therefore, the challenge is to refine the monitoring program such that it serves both of these objectives at minimum cost and without causing undue stress on the outmigrating smolts.

An effective smolt monitoring program requires adherence to the following principles:

(1) the purpose(s) of each monitoring exercise or experiment must be clearly defined;

(2) the techniques which are to be applied, including experimental design, equipment, and analytical methods, must match the stated purpose of the exercise; and

(3) there must be a commitment of sufficient resources and effort to complete the exercise.

Any monitoring program which might be proposed will reflect a tradeoff among effectiveness, cost, and potential interference with the migration. For example, increased use of remote sensing technologies such as PIT tags and hydroacoustics can reduce the handling stress which is associated with monitoring, but greatly increase equipment costs. Furthermore, although investment in research and monitoring programs does not guarantee practical, effective results, such investment may pay off many-fold. Responsible parties need to make decisions about the amount of money and effort that can be devoted to monitoring and research. The recent history of smolt monitoring in the Columbia Basin suggests that fishery managers are reluctant to experiment with alternative mitigation approaches, because such experiments may endanger smolts. However, we believe it is worthwhile to consider a new strategy to improve research efforts and mitigation evaluations.

A NEW STRATEGY FOR SMOLT MONITORING

Step 1: A Shift in Priorities

The first step in improving the smolt monitoring program should be to assign a higher priority to research. The existing evidence supports neither a contention that research is “too risky” nor a position that current flow management practices maximize smolt survival. For example, work by Miller and Sims (1984) questions the validity of current management practices by demonstrating that consistent, robust relations between flow or spill fractions and survival do not exist. The survival estimates and accompanying management strategies produced to date, while interesting, appear to be insufficiently rigorous to allow even pairwise comparisons, were those appropriate. Considering that flows and spills and other factors have varied while these experiments have been in progress, there is very little that can be said about how the management of these factors affects survival. It appears counter-productive to object to increases in tagging experiments for fear of "damaging the run" or to refuse to use management capabilities to support monitoring experiments on the basis that particular flow and spill patterns are "required" to enhance the run.

This is not to suggest that research should proceed recklessly. But the search for a better understanding demands commitment. No research worth doing will be totally without risk of loss to juvenile fish, but a program without research may preclude opportunities to enhance the production and survival of juvenile fish.

Step 2: Better Use of Technology

Given that the amount of resources that can be devoted to monitoring and evaluation are limited, the key to better understanding of downstream migration is to maximize the amount of information that is obtained from each monitoring exercise and from each fish which is marked or handled in any way. This involves not only the use of improved technologies, but also closer attention to experimental design and better coordination of effort among the involved research and management agencies. A successful research effort needs a well-conceived experimental design or a set of alternative designs, with cost estimates; an agreement on which design(s) should be implemented; and a commitment to complete the experiments.

Experiments to estimate survival seemingly have failed in the past, because of a combination of insufficient recovery of marked fish and little or no replication of experiments. The percentage of marked fish recovered and the number of experimental replicates have to be increased, in order both to provide more precise estimates of survival parameters and to reduce the ranges of the implied confidence intervals to acceptable levels. Since huge increases in the numbers of marked and released fish are unlikely, the only alternative for obtaining better information is to increase the recapture rate of marked fish.

For conventional marking techniques such as freeze-branding, which requires up-close visual inspection to read the marks, an increase in the recapture rate implies handling increased numbers of fish passing the sampling station. This is appropriate only if the fish will be passing the sample point in high concentrations; that is, they are sampled (recaptured) at short distances from their release site and within a short time following release. The marked fish can not have time to disperse, or an enormous number of fish will have to be handled to get a sufficient number of marked fish. Visual marks should be adequate for experiments which are routinely done to estimate sampling efficiencies at particular dams. In such experiments, known numbers of marked fish are released within a few kilometers upstream of the dam and recaptured at the dam. Large numbers of fish are not required and replicates can be obtained by using unique marks.

More extended studies, which might involve recapture at a number of dams over a longer period of time, should take advantage of remote sensing techniques such as PIT tags. These techniques do not require the removal of fish from the bypass systems. Although PIT tags are expensive and require specialized monitoring equipment, the potential gains in information which could accompany widespread application of this technology warrant a substantial investment in their development and use. In time, the unit cost per application should decline and the efficiency of the technology should improve.

Other technologies such as hydroacoustics and radio tagging have their place in the smolt monitoring program, and efforts to improve these techniques should continue. Radio tags are especially useful in providing detailed information about the movements and fates of individual fish. Hydroacoustics seems to be most productive when one is interested in the mass movements of large numbers of fish. Researchers should use these technologies in situations to which they are best adapted, such as in the investigation of how the timing of spills at individual dam sites affects survival.

Step 3: Statistical Methods

Statistical methods were discussed above, so only a few points need to be reiterated here. The application of advanced monitoring equipment combined with the completion of bypass facilities systems at most major dams in coming years should produce a wealth of monitoring data. Researchers can draw strong inferences from these data by applying powerful statistical techniques such as the release-recapture protocols proposed by Burnham et al., and should pursue their use. The stochastic reservoir passage model discussed in Chapter 4 provides a novel way of looking at reservoir passage and investigators should utilize this model as well. To permit informative statistical analyses, data must be collected within strict formal guidelines set forth by sound experimental designs. Statisticians have been and must continue to be involved in the research planning from the beginning.

Step 4: The Role of Models

The problem of assessing survival of migrating smolts is much more complex than previously imagined. This complexity has frustrated many efforts to investigate survival, since researchers have either failed to appreciate the reticular nature of the problem or simply were unable to deal with it effectively. As discussed above, poor understanding of error structure and questionable assumptions which lead to faulty experimental designs have been especially troublesome. Years of experience in the Columbia River Basin have given scientists in the region a clearer picture of the research problem, but they remain stymied in resolving several major issues.

One would like to be able to try a wide variety of experiments within the Columbia River Basin to estimate smolt abundance and survival, and then stay with those experimental designs that appear to be most effective. However, such an approach consumes time, costs a lot of money, wastes resources, and provides no guarantee of success. The history of smolt monitoring in the Columbia proves this.

An alternative to repeated, real-world experiments is to experiment with a model of the system as part of the design phase to develop a monitoring and research strategy. To date, no one has taken this tack on a systemwide scale in the Columbia. Two separate simulation models of downstream passage in the Columbia River Basin have been developed--the FISHPASS model developed by the Corps of Engineers (Tanovan 1985) and the

System Planning Model developed by the Northwest Power Planning Council (Webb et al. 1986). (The System Planning Model actually simulates the entire salmon life cycle, but has a downstream passage component within it.) Both of these models were developed primarily for evaluating the relative impacts on fish survival of alternative management actions, including altered flow and spill regimes and transportation of juveniles. Although the models perhaps could be extended to incorporate uncertainty, their existing structure and deterministic nature currently makes them unsuitable as a research planning tool within the framework envisioned here.

In order for a model to be useful in the present context, it explicitly must incorporate stochastic features and include a representation of the sampling process. The role of this model would be to test the effectiveness of proposed monitoring schemes and survival experiments, in terms of accuracy and precision of parameter estimates that might be expected under varying conditions. These conditions would be defined such that they mimic real-world constraints and uncertainties.

The premise for proposing the development of a comprehensive sampling model is that the Columbia System is too complex, with too many variables and interactions among components, to allow for an analytical solution to questions of sample size, replication, and distribution of sampling effort. One can make simplifying assumptions for given experimental approaches and estimate survival and mortality parameters, but experience suggests that such estimates can be quite misleading. It may be possible to test how such estimates will hold up in the real world, and to gain an insight into their robustness, by using simulation to set up and test various scenarios. To some extent, the RELEASE program described earlier can accommodate this type of sensitivity analysis for release-recapture experiments. However, although this may be sufficient for single experiments, it does not provide the flexibility and muscle necessary to do a full-scale, integrated analysis of all migration routes.

A comprehensive simulation model would allow one to bring into the analysis many of the confounding problems in the real world (e.g., differential survival, the influx of non-targeted populations, fluctuations in flows and spills, transportation, etc.). It would incorporate many of the system complexities that simpler statistical models assume away. The goal of the modeling exercise would be to determine if it is possible to design a real-world experiment, with real-world environmental and management confounding factors, to test the hypotheses of interest. If the modeling could help demonstrate that one mitigation approach is more robust than another of comparable or greater cost, the modeling exercise would be time and effort well spent.

The knowledge exists to build a model which has the desired features. Existing passage models such as FISHPASS provide the basic conceptual structure of the system. In order to suitably modify FISHPASS it would be necessary to:

(1) incorporate random number generators than can generate parameter values from a number of alternative distributions, thus adding a more realistic uncertainty to the simulated system;

(2) add a representation of the alternative sampling processes that are possible at each dam;

(3) include alternative release sites for release-recapture experiments; and

(4) modify the reservoir passage component of the model such that it more accurately captures the stochastic nature of reservoir passage.

It may be better to start anew in building such a model, since changing the source code of FISHPASS to include these modifications would be a laborious task. Such a model-building effort would use the FISHPASS algorithms as a blueprint. The new model would retain those features of FISHPASS which address the objectives of the newer model and omit those features that are superfluous.

One of the often-encountered problems of complex models is that they are difficult to modify once they have been built. Steps should be taken to ensure that this does not occur in the model proposed here. It is important to be able to modify the model to reflect changing conditions in the real system, such as the completion of bypass systems, and to be able to update the model as understanding of the system improves. Programming techniques such as modular programming can facilitate such modifications.

CONCLUSIONS

Monitoring of the outmigration and evaluation of mitigation strategies will continue to be a major concern of fishery managers and researchers in the Columbia River Basin for some time. In this chapter, we have discussed some of the problems encountered in smolt monitoring and introduced some of the potential technological and statistical advances which may alleviate some of these problems. We believe that some of these problems can be resolved. However, their resolution will require a sound research design, a commitment on the part of fisheries managers and funding agencies, innovation and discipline on the part of researchers, and cooperation all around.

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GLOSSARY

adaptive management – an approach to reducing uncertainty by using management actions as experiments which provide information about the system. Such information is then used for more efficient management.

adult – for salmonids, a fish that is sexually mature.

alevin – a newly-hatched salmon or trout prior to absorption of the yolk sac.

anadromous – fish which spawn in fresh water but spend a significant portion of their life in the ocean.

Beverton-Holt curve – a spawner-recruit relationship characterized by a curve depicting the number of recruits increasing to an asymptotic limit as the number of spawners increase.

brood year – the year in which a fish begins life.

cohort – fish offspring of the same brood year.

escapement – the number of adults which survive to reach the spawning grounds.

fallback – a situation in which fish which have ascended a dam are swept back downstream of the dam.

fecundity – usually refers to the number of eggs produced per female.

fingerling – a young or small fish, larger and more developed than a fry.

fishery – the complex of interactions among a fish population(s), the people which exploit them, and the environment.

fry – young, recently hatched fish generally capable of feeding only on microorganisms.

heuristic model – a model which serves primarily as an experimental device for exploring modeling techniques.

hierarchy – an arrangement of objects into a graded series based on the relationships among the objects themselves.

jack – a precocious male salmon or steelhead trout, generally fish that have matured at least one year earlier than most members of the same cohort .

juvenile – fish that are not sexually mature.

juvenile production – generally used in this text to refer to the production of smolts.

life stanza – a distinct period in the salmonid life cycle.

module - a simulation model which can operate independently or in tandem with other modules.

outplanting - placing fry or fingerling into areas for rearing to supplement or replace natural reproduction.

parr - juvenile anadromous salmonids which normally reside for a year or more in fresh water and are not capable of tolerating saline water. May refer to steelhead trout, coho, sockeye, or chinook salmon.

pre-smolt - similar to parr.

recruit - fish which are newly joined with a population under consideration.

recruitment - the addition of new members to the aggregate population under consideration.

redd - the spawning nest of salmonids; usually a scooped depression in clean gravel in which eggs are deposited and buried.

resolution - the ability to distinguish between two separate objects.

Ricker curve - a spawner-recruit relationship characterized by a dome shaped curve.

run size - as defined by the Council, the total number of fish returning to the mouth of the Columbia plus ocean harvest.

run year - the year in which a fish returns to spawn as an adult.

scope - the relative temporal and spatial extent of the system under consideration.

seeding rate - an index used to reflect the number of fry (or potential fry which can be produced by the available spawners) present in a stream relative to the maximum number which can be supported by the available habitat.

shaker - a fish which is smaller than the legal size limit for harvest but which is susceptible to the angling gear..

smolt - a juvenile salmonid which is physiologically prepared to outmigrate from fresh to saline waters.

smoltification - the physiological process which prepares an anadromous fish for life in saline waters.

spawner-recruit curve - the relationship between some measure of biomass present in spawning adults and the biomass of recruits derived from the spawning adults.

stock - a population of fish which remain genetically, spatially, or behaviorally separated from other populations and which shares a common life history among its members.

systems analysis – a body of theory and techniques used to understand complex systems, usually involving advanced mathematical and statistical techniques and the use of computers.

water budget – a program to provide addition instream flows during the period of peak outmigration to increase the travel rate of migrating smolts.

PART II

MODELS FOR COST-EFFECTIVENESS ANALYSIS

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Part II

Models for Cost Effectiveness Analysis

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PREFACE TO PART II

This part concerns the development of methods for analyzing the cost and long-term fish production implications of alternative management strategies for enhancing the anadromous fisheries in the Columbia River Basin. The methods presented in this part are designed to address both the general requirements of and the individual tasks specified in the agreement between the Bonneville Power Administration (BPA) and Resources for the Future (RFF) that pertain to the development of a systems model which would allow simulating the fish production effects of alternative protection, mitigation, or enhancement strategies for the purpose of comparing the cost-effectiveness of such alternatives. Such strategies include, among others, long-term changes in the amounts and locations of water diversions and long-term changes in the instream flow regime.

The methods presented in this part also are designed to support the requirements of the Pacific Northwest Electric Power Planning and Conservation Act of 1980 (Act) that pertain to the adoption of cost-effective fish mitigation measures for achieving each sound biological objective ultimately to be specified by the Northwest Power Planning Council (Council). The direct language of that Act stipulates that where there are several equally effective means of achieving the same sound biological objective, the alternative with the minimum economic cost must be employed. Thus, when two or more measures can achieve the same mitigation function, cost-effectiveness analysis can be used to identify the preferred option. Moreover, where the biological objective is specified in terms of adult fish production, such as the total number of adult fish that are harvested in the ocean and that escape the ocean and return to the Columbia River either to be harvested or to spawn, the cost-effectiveness analysis must embrace a long-term, system-wide approach. The reason for this is that adult fish production requires analysis of the entire fish life-cycle and this takes place over the whole system.

There is nothing in the direct language of the Act that expressly prohibits the use of more conventional methods of economic analysis, such as cost-benefit analysis, in analyzing and comparing alternative management strategies. This probably was intentional rather than an oversight, especially if the drafters of the legislation viewed economic analysis in the context of mitigation in the Columbia River Basin more properly as an evaluation tool rather than as a decision rule. Economic efficiency is important but it is not the only consideration in decisions on fish mitigation in the Columbia River Basin. Several other considerations also bear on these decisions, such as the distribution of the costs and benefits of mitigation and the welfare of those who live in the Pacific Northwest. Nonetheless, economic efficiency should not be excluded entirely. Estimates of benefits and comparisons of costs and benefits can be helpful in decisions on mitigation and should be included along with other measures of the effects of proposed management strategies in the information upon which decisions are based.

The methods presented in this part are designed to support two kinds of cost-effectiveness analyses. Each kind of analysis requires a different method or approach. The first kind of cost-effectiveness analysis is an assessment and comparison of the cost and fish production implications of a prespecified set of alternative mitigation strategies. The appropriate method to use for this kind of analysis is simulation. An economic-ecologic simulation model designed to evaluate alternative mitigation strategies in the Columbia River Basin is outlined in chapter 2. The second kind of cost-effectiveness analysis involves the design of mitigation strategies by systematically choosing from a menu of mitigation measures. The appropriate method to use for this kind of analysis is mathematical programming. A mixed-integer linear programming model designed to identify cost-effective mitigation strategies for the Columbia is outlined in chapter 3. Methods for estimating the benefits of mitigation are not presented in this part because the development of such methods lies beyond the scope of the research specified in the agreement between BPA and RFF.

A system-wide approach was taken to the development of cost-effectiveness models at this stage in planning the research even though gaps in data availability and in understanding of key interrelationships currently exist. There are several reasons for contemplating a system-wide model at this stage:

- to identify the principal components of the overall model, guide the design of those components, and identify the linkages among components;
- to ensure consistency in the temporal and spatial resolutions of data and information both within the individual components of the overall model and in the linkages among components;
- to help identify sources of uncertainty, gaps in data availability, and gaps in current knowledge needed to undertake a system-wide, cost-effectiveness analysis; and
- to help identify, organize, and rank research needs.

Experience with the design, development, and application of large scale environmental models involving multiple disciplines teaches a lesson. The individual components of large scale models cannot be expected to fit together unless they are designed from the outset to be integrated. The discussion in this part takes this point as a keystone.

Decisions on fish protection, mitigation, and enhancement in the Columbia River Basin will be made with or without the availability of models to assess the cost-effectiveness of alternative management strategies. Although they are not a substitute for all the information that will be needed for such decisions, models can be useful. If they are used properly and if they are systematically improved as more data and more knowledge of key interrelationships become available, they can improve significantly the quality of the information available for decisions. The methods outlined in this part represent the beginning of a process which aims to improve the quality of information available to decision makers in the Pacific Northwest on the economic and fish production implications of alternative management strategies for the Columbia River Basin.

Chapter 1

Economic Analyses of Fish Mitigation Strategies

INTRODUCTION

This part is a report on methods for analyzing the cost and long-term fish production implications of alternative management strategies for improving the anadromous fisheries in the Columbia River Basin and for identifying cost-effective alternative means for achieving biological objectives ultimately to be specified by the Northwest Power Planning Council (Council).

The methods presented in this part are designed to support two kinds of cost-effectiveness analyses. The first kind of analysis is an assessment and comparison of the cost and fish production implications of a prespecified set of alternative mitigation strategies. The appropriate method to use for this kind of analysis is simulation. The second kind of analysis involves the design of cost-effective fish mitigation strategies by means of a systematic search procedure. The appropriate method to use for this kind of analysis is mathematical programming. Comparisons of the costs of alternative mitigation strategies that achieve the same level of fish production, analyses of the tradeoffs between levels of fish production and the costs of mitigation, and identification of cost-effective fish mitigation strategies are referred to collectively as cost-effectiveness analyses.

The methods presented in this part are designed to address a different set of issues and problems than those presented in Part I, although the methods, issues, and problems are closely related. The methods in Part I attempt to quantify the fish production implications of particular mitigation measures and system-wide mitigation alternatives. The methods in this part attempt to quantify both the fish production implications of mitigation alternatives and the costs of those alternatives. Thus, the methods in Part I are concerned principally with biological and probabilistic aspects of estimating levels of fish production. This type of information and modeling is essential for assessing the effectiveness of particular mitigation strategies in achieving stated biological objectives. Part II builds on this foundation and is concerned primarily with estimating the costs of those strategies and with exploring tradeoffs between levels of fish production and the costs of mitigation. The fish life-cycle simulation model(s) required for the cost-effectiveness analyses described in this part will be developed as part of the research proposed in Part I.

This chapter is organized in six principal sections. The first section defines some terms used in the chapter. The second section presents the different types of cost-effectiveness analyses and assesses their applicability to the analysis of fish mitigation in the context of the Columbia River salmon and steelhead fisheries. The third section describes the conditions that were assumed for the design and development of methods of cost-effectiveness analysis presented in this part. These conditions include the biological objectives for the fisheries to be established by the Council, the economic criterion used to compare mitigation alternatives, the types of costs included in the cost-effectiveness analyses, and the nature of the policy constraints imposed on the analyses. The fourth section describes three problem types likely to be encountered in assessing the cost-effectiveness of alternative mitigation strategies. The fifth section briefly presents the two cost-effectiveness models proposed for development. The last section presents an overview of the remaining chapters in this part of the Phase II report.

The methods outlined in this part of the report are intended to be used to improve the quality of information available to decision makers in the Pacific Northwest on the economic and fish production implications of alternative mitigation strategies for the Columbia River Basin. A few words on the scope of the economics portion of the research may help place these methods in perspective.

First, this is a report on the design of studies for fish mitigation policy and not a report on a fully developed set of methodologies. Although considerable investigation was necessary in order to develop a comprehensive, system-wide analytical framework for cost-effectiveness analyses of mitigation strategies for the Columbia, the methods presented in this part are not fully formulated. They were developed only to the extent necessary: (1) to determine if it would be feasible to apply proposed methods to the analysis of mitigation strategies in the Columbia River Basin, and (2) to determine what would be required by way of data, simulation models, computer software, and computer hardware to fully develop and to apply those methods.

Second, the methods outlined in this part of the report are designed to analyze the cost-effectiveness of alternative mitigation strategies. They are not designed to estimate the economic benefits of strategies or to compare the costs and benefits of strategies. The development of methods for estimating the benefits of fishery enhancement lies beyond the scope of the research specified in the agreement between the Bonneville Power Administration (BPA) and Resources for the Future (RFF).

Third, the methods outlined in this part to analyze the cost and long-term fish production implications of alternative mitigation strategies pertain only to the salmon and steelhead fisheries. They do not address other fish and wildlife concerns in the Columbia River Basin. Because of this focus, the biological part of the analysis and some aspects of the hydrosystem comprise only a portion of the Columbia River Basin, together with the ocean fishery: from Grand Coulee Dam on the mainstem Columbia, Hells Canyon Dam on the mainstem Snake, and Dworshak Dam on the Clearwater to the mouth of the Columbia River at the Pacific Ocean. Upstream areas in the states of Washington, Oregon, Idaho, Montana, and Wyoming, and in Canada, with the exceptions of storage and hydropower generation, are not included in the analyses.

DEFINITION OF TERMS

Several terms used in this chapter require definition. Fish mitigation measures refer to structural and nonstructural means for enhancing fish production. Examples of structural measures include fish hatcheries, by-pass facilities at dams for the passage of juvenile fish, and transportation of juvenile fish by barge and truck to the estuary below Bonneville Dam. Examples of nonstructural measures include additional spills at dams and flow enhancement of the Columbia and Snake rivers, both for the benefit of the anadromous fisheries.

Two other terms that require definition are management alternative and management strategy. A management alternative is any individual measure used to enhance fish production. A management strategy is a set of fish mitigation measures used to enhance fish production.

Another pair of terms that require definition are optimization and least-cost set of alternatives, or strategy. Optimization refers to the analytical process of maximizing or minimizing an objective such as maximizing the number of total adult fish (or fish biomass) subject to the availability of funds for mitigation in the basin, or minimizing the costs of mitigation to meet a particular biological objective specified by the Council. A least-cost strategy refers to that combination of mitigation measures that can achieve a particular biological objective at the lowest possible cost, subject to a set of technical constraints and a set of administrative, legal, and political conditions. For most public policy applications, there are several least-cost strategies, one for each set of administrative, legal, and political conditions assumed for the analysis. Thus, least-cost is used to describe strategies that are efficient within broad public policy objectives established for the region rather than strategies that are based narrowly on technical feasibility and economic decision criteria alone.

Simulation is the process of simulating, or mimicking, the behavior of a natural, engineering, or economic system using a mathematical model called a simulation model. For the methods presented in this part, simulation is used synonymously with mathematical procedures for mimicking the behavior of the hydrosystem and the anadromous fishery. Optimization is used synonymously with mathematical procedures for systematically searching for cost-effective management strategies.

COST-EFFECTIVENESS ANALYSES

Cost-effectiveness analyses have been used by engineers, economists, and public administrators for many years to inform decisions on public works projects and other public programs. They provide those responsible for decisions with a quantitative guide for comparing and ranking alternatives in situations where monetary measures of output are not available but where nonmonetary measures of effectiveness can be established. Cost-effectiveness analyses are ideally suited to the problem of mitigating hydropower impacts in the Columbia River Basin. Moreover, they are required by the Pacific Northwest Electric Power Planning and Conservation Act (Northwest Power Planning Council, 1987b). The appropriate measure of effectiveness to use in these analyses is the biological objective or objectives ultimately to be specified by the Council (see discussion below). The appropriate costs to use in these analyses are the economic costs of mitigation measures.

Forms of Cost-Effectiveness Analysis

Cost-effectiveness analysis assumes various forms. The different forms need to be distinguished because they have important implications both for the cost-effectiveness analysis and for the development of methods. In its simplest form, the analysis is a comparison of the costs of strategies that achieve the same level of “effectiveness”. In comparing two strategies, the strategy with the lower cost is said to be more cost-effective than the strategy with the higher cost. In comparing a set of alternative strategies, the strategy with the lowest cost is said to be the most cost-effective.

Sometimes it is not possible, or practical, or even desirable to identify strategies that achieve the same level of effectiveness. There are several reasons for this, but three are particularly relevant to the analysis of management strategies for the Columbia River Basin. The first reason is that the method used to assess the cost and effectiveness of particular management strategies may not be capable of ensuring the same level of effectiveness (target levels of adult fish production at different times throughout the planning period) across all the strategies to be compared. For example, there may exist some combination of hatcheries and by-pass facilities, on the one hand, and some combination of hatcheries and spills, on the other, capable of achieving the same level of effectiveness but the method used may incorporate no systematic search procedure for identifying them. This is a problem of analysis. Lack of such a procedure is a weakness of simulation and its presence is a particular strength of mathematical programming. (The systematic search procedure in mathematical programming used to identify alternative management strategies that achieve the same level of effectiveness is not the same systematic search procedure used to identify the combination of mitigation measures that achieve the same level of effectiveness at the lowest possible cost. They are different procedures. In mathematical programming, the first search procedure is used to find an initial feasible solution. The second search procedure is used to find the optimal solution among all possible feasible solutions.) The two principal methods of cost-effectiveness analysis--simulation and mathematical programming--are discussed in more detail later on in this chapter and in considerable detail in chapters 2 and 3, respectively.

The second reason for differences in the levels of effectiveness across all the management strategies in the analysis pertains to the case of multiple measures of effectiveness, such as the number of adult fish of different stocks rather than the total of all adult fish. In such cases, the management strategies in the analysis may affect the different measures of effectiveness differentially. Some measures of effectiveness may increase while other measures decrease, and consistency across all the strategies in the analysis can be problematic. This is a technical problem of jointness, not a problem of analysis. This condition is especially troublesome if the number of measures of effectiveness in the analysis is large. In such cases, multiple objective analysis can be used. Jointness is a problem for all environmental cost-effectiveness analyses which involve ecological systems and multiple measures of effectiveness. It will have to be addressed in analyzing the cost-effectiveness of alternative management strategies for the Columbia.

The third reason for differences in the levels of effectiveness across all the strategies considered in the analysis is intentional. In some situations it may be desirable to explore cost-effectiveness tradeoffs rather than to assess the cost-effectiveness of strategies that achieve the <u>same</u> level of effectiveness. For example, the cost of achieving the next increment of returning adult fish may be relatively modest in which case the region acting through the Council may find it desirable to raise the biological objective (target level of adult fish production). Or the cost of achieving the last increment of adult fish may be deemed excessive in which case it may be desirable to lower the target level of returning adult fish. The cost-effectiveness tradeoff analysis provides the information on both the costs and the effectiveness of mitigation (level of adult fish production) that is needed in order to inform that decision.

In situations where the levels of effectiveness are not the same across all the mitigation strategies in the analysis, costs cannot be used alone to compare and to rank strategies. Rather, the costs and corresponding levels of effectiveness must be compared as sets of attributes, in multiple dimensions. In these situations, it is sometimes helpful to “normalize” the costs of alternative strategies for purposes of comparison. Average costs and marginal costs are examples of such normalizations.

In the case of a single measure of effectiveness and a single measure of cost, the comparison of mitigation strategies can be made in two dimensions--a cost dimension and an effectiveness dimension. In such cases, it is often instructive to plot the costs and the effectiveness of the different mitigation strategies on a graph. Such a visual aid is helpful in separating the more "cost-effective" strategies from the less "cost-effective" strategies. It is also helpful in decisions on the level of the biological objective in those situations where the biological objective has yet to be established. Such an analysis is referred to in this part as cost-effectiveness tradeoff analysis.

An example of a cost-effectiveness tradeoff analysis for the case of a single measure of effectiveness and a single measure of cost is shown in Figure 1.1 for fish mitigation in a typical river basin. The measure of cost in this figure is the total cost of a particular mitigation strategy. The measure of effectiveness is the total number of adult fish that are available for harvest and for spawning. The curve in this figure represents the set of least-cost mitigation strategies for achieving different target levels of adult fish production. The strategies represented by this curve are the most cost-effective strategies possible in the sense that there are no other strategies that can meet the same target levels of adult fish production at less cost (subject to the conditions imposed on the analysis). Also shown in this figure is a set of less cost-effective mitigation strategies. These strategies are represented by the symbol "+" and lie above the cost-effective curve. For a particular target level of adult fish production, these strategies cost more than those that lie on the cost-effective curve and thus they are less cost-effective. The slope of the curve in Figure 1.1 at any point represents the marginal cost of mitigation (the cost of the next increment of mitigation) in this typical river basin.

With multiple measures of effectiveness, the cost-effectiveness tradeoff analysis becomes more complex. This is especially true if there are many measures of effectiveness in the analysis and if these measures move in opposite directions as the total cost of mitigation increases (or decreases). For example, the production of adult fish of one stock may increase while at the same time the production of another stock decreases. To ensure consistency in the measures of effectiveness in the cost-effectiveness analysis, and thus comparability across alternative management strategies,

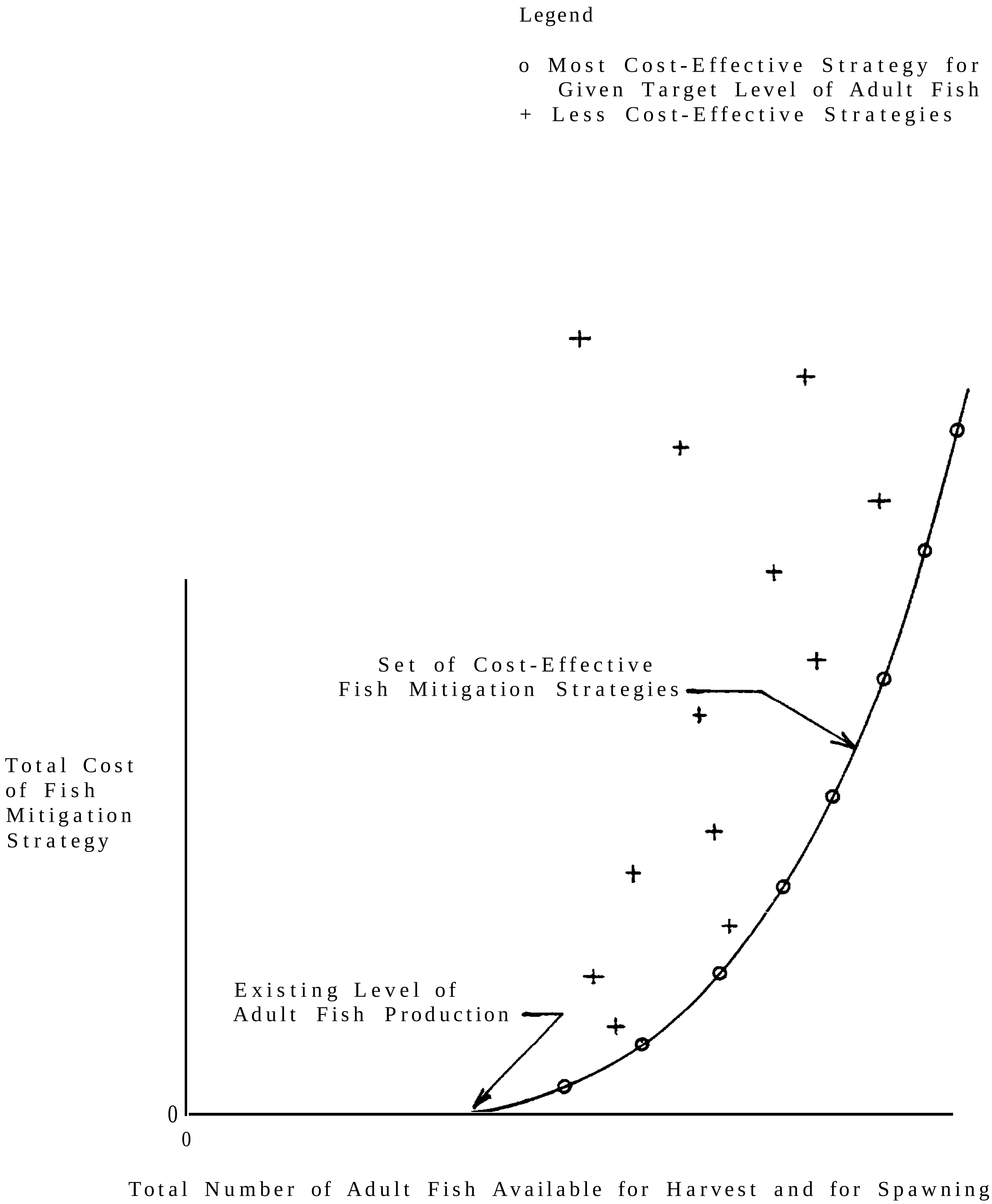

Figure 1.1. An Example of a Cost-Effectiveness Tradeoff Analysis for Fish Mitigation in a Typical River Basin

mathematical programming is often required. One of the particular strengths of mathematical programming is its ability to ensure consistency in the measure (or measures) of effectiveness across the management strategies to be compared, with the exception of the jointness problem discussed above.

It is likely that multiple measures of effectiveness will be involved in analyses of management strategies for the Columbia River Basin. These measures include the number of juvenile fish of particular species produced in a given subbasin, the number of juvenile fish of particular stocks that safely reach the estuary below Bonneville Dam, the number of adult fish of particular species in the ocean, the number of adult fish of particular species that escape the ocean and enter the lower Columbia River, the number of adult fish of particular species in river segments traversed by more than one stock of the same species, and the sizes of adult fish runs of particular stocks at particular locations in the river system. Because of this multiplicity of possible measures of effectiveness, the different forms of cost-effectiveness analysis described above will be helpful in analyzing the cost-effectiveness of particular management strategies.

Types of Models Used for Cost-Effectiveness Analyses

There are two fundamentally different kinds of models applicable to the analysis of the cost-effectiveness of alternative mitigation strategies in the context of the anadromous fisheries in the Columbia River Basin. One kind of model is a simulation model. (See the definitions of simulation and simulation models in the first part of this chapter.) The other kind of model is a mathematical programming model. A simulation model can be used to assess the cost and fish production implications of particular management strategies. It also can be used to search for a set of management strategies that meet the same level of effectiveness (target level of adult fish production) or to search for the most cost-effective strategy, although unlike a mathematical programming model it contains no formal search procedure and in practice this use of a simulation model often becomes impossibly burdensome. More will be said about this below.

Simulation Model. When a simulation model is used for the cost-effectiveness analysis, a management strategy called a "scenario" is specified completely. In the case of the Columbia, the scenario is a set of protection, mitigation, or enhancement measures. There are no management decisions (choice variables) in the analysis. The analysis involves simulating the effects of the scenario and assessing the outputs of interest--the cost of the strategy, the number of fish reaching maturity in the ocean, and the number of adult fish of a particular species escaping the ocean and entering the Columbia River to be harvested or to spawn. A number of scenarios are developed and analyzed, and the outputs of interest are compared. The management strategies are ranked according to the criterion adopted for the analysis, in this case the cost of meeting a target level of adult fish of a particular stock or species. In this approach, identification of the most cost-effective strategy is not guaranteed, as noted above, and exactly meeting a particular biological objective is unlikely as well. In fact, identification of the least-cost combination of mitigation measures for a particular biological objective would be highly unlikely.

For large, complicated problems such as fishery enhancement in the Columbia, it would be virtually impossible to identify the most cost-effective combination of structural and nonstructural measures using simulation. The set of results shown in Figure 1.1 denoted by the symbol “+” is typical of the results obtained from simulation models--even in relatively simple cases where only one measure of effectiveness is involved. However, in contrast to no model at all, cause and effect type information as well as the cost and fish production implications of particular management strategies are available for planning decisions.

Mathematical Programming Model. Mathematical programming is useful in the design of strategies where the goal is to maximize or minimize a criterion function, such as the cost of fishery improvements. A mathematical programming model can, if successfully developed, be used to assist in identifying strategies that achieve the same level of effectiveness at the lowest possible cost, for the system as a whole or for a subdivision of it.

When mathematical programming is used for the analysis, the particular management strategies to be assessed are not specified *a priori*. Rather, mathematical programming is used to identify that combination of potentially available fish hatcheries, levels of improvements in natural spawning and rearing areas, operations of dams and reservoirs, improvements in river flows, and other measures that can achieve the biological objective at the lowest possible cost, subject to limits, such as firm power requirements, called constraints. This approach often requires complex programming techniques which go beyond those needed for simulation.

Comparison of Modeling Approaches. In addition to differences in the computational complexity of the mathematical programming and simulation modeling approaches, there may also be differences in the accuracy of their outputs. Because mathematical programming models tend to grow large in size (measured by the number of management and state variables and by the number of constraining relationships) and thus to become unmanageable, and in some cases even "unsolvable", simplifying assumptions are often required. These simplifying assumptions can affect the accuracy of the results obtained from these models.

Assumptions made to reduce the size of mathematical programming models are generally not required of simulation models. Thus, simulation models are able to provide more accurate assessments of the cost and fish production implications of particular management strategies than programming models. Therein lies the dilemma for cost-effectiveness analyses. Simulation models are able to provide more accurate assessments of the cost and fish production implications of particular management strategies, but they generally fail to identify the least-cost strategy. Mathematical programming models can identify the least-cost strategy (subject to the conditions imposed on the analysis), but due to the simplifying assumptions that are generally required, they do not mimic reality with the fidelity of simulation models.

Although each approach has desirable features, neither entirely satisfies the needs of the Columbia River Basin fish mitigation analyses. This suggests a combination of the two approaches.

One way to utilize the strengths of both approaches is to use mathematical programming as a "screening" device to assist in identifying a set of technically and politically feasible management strategies that achieve the same level of effectiveness or that have desirable cost-effectiveness properties, or both. This set of strategies can then be simulated using the more detailed simulation model. The latter will provide more accurate estimates of the mitigation costs and more accurate estimates of the fish production implications of the different mitigation strategies. These more accurate estimates of costs and of the numbers of adults available for harvest and for spawning can then be used to identify the most cost-effective management strategy (see Figure 1.1).

Mathematical programming models are more difficult to build than simulation models because of the many approximations and adaptions that are required in order to formulate (and in some cases reformulate) a problem that can fit within one of the standard mathematical programming structures (e.g., linear program, mixed-integer linear program, nonlinear program). In addition, there is far less experience with the use of mathematical programming models in this type of application than with the use of simulation models. Therefore, the development of the mathematical programming model is considered experimental at this stage in the research. Nonetheless, an attempt should be made to develop a mathematical programming model that can be used in conjunction with the simulation model.

An economic-ecologic simulation model designed to assess the cost and fish production implications of particular management strategies is outlined in chapter 2. A mathematical programming model designed to identify the least-cost combination of mitigation measures for a particular biological objective and set of administrative, legal, and political conditions is described in considerable detail in chapter 3.

Analyzing Uncertainties in Cost-Effectiveness Analyses

The discussion of methods for analyzing the cost and fish production implications of alternative management strategies up to this point in the chapter has assumed complete certainty in the analyses. However, such a discussion would not be complete without mentioning the potential uncertainties in the outputs of the analyses and without describing approaches that might be used to analyze the nature and magnitude of these uncertainties. The Columbia River Basin is too large and the fishery management strategies are too complex to ignore this. (See also the discussion of uncertainty in Part I.)

For purposes of the cost-effectiveness analyses described in this part, uncertainty is defined as the possibility of differences between *ex ante* projections and *ex post* outcomes (Spofford, Krupnick, and Wood, 1986). Of course, *ex post* outcomes are not available at the time of the *ex ante* analysis. Thus, a more practical measure of uncertainty is variability in projections based on analysis. Such variability can be caused by stochastic, natural variations (e.g., streamflow) that can be described *ex ante* by known (objective) probability distributions, and it can be caused by factors where information on variations currently does not exist. Differences between projections and outcomes also can be caused by factors that are unknown at the time of the *ex ante* analysis. This source of uncertainty is called “surprise” in the water resources literature and includes such events as droughts that have never been experienced before and diseases in anadromous fish populations that are currently unknown.

Much of the uncertainty in *ex ante* projections is inherent and cannot be reduced either with more data or with more research. However, some of the uncertainty in these projections is caused by lack of data or a poor understanding of basic interrelationships, or both, and can be reduced with more data and more research. For purposes of research planning and data collection, it is important to distinguish between these two categories of uncertainty, and to identify those sources where the uncertainty can be reduced and those sources where it cannot.

Uncertainties in projections derive from the compounding of uncertainties spread throughout the analysis. There are three principal kinds of uncertainty--model structure uncertainty, parameter value uncertainty, and input data uncertainty. All three kinds can affect ex ante projections of fish production, estimates of costs, and measures of the cost-effectiveness of particular management strategies.

There are two basic approaches to analyzing uncertainty. The first is error analysis. This is the more formal of the two approaches and requires mathematical functions with continuous first and second derivatives. The second approach is Monte Carlo simulation. This approach is not as formal as error analysis and requires neither continuous functions nor derivatives of those functions. Monte Carlo simulation is the only practical approach to use to analyze the uncertainties in projections of fish production, estimates of mitigation costs, and measures of the cost-effectiveness of mitigation strategies.

Because of the implications of uncertainties in model projections for being able to identify cost-effective mitigation strategies for the Columbia River Basin, the uncertainties in model outputs should be analyzed. Such an analysis can be made by placing the economic-ecologic simulation model outlined in chapter 2 within a Monte Carlo simulation framework. This use of the simulation model should be considered in developing the model. The mathematical programming model described in chapter 3 also could be placed within such a simulation framework, but this has not been done before for a problem the size of the Columbia, or even one of its subbasins, and therefore it should be considered experimental.

The capability to analyze levels of uncertainty in projections of fish production, estimates of mitigation costs, and measures of the cost-effectiveness of management strategies should be developed. However, the models required for this analysis should be developed in a logical sequence, beginning with deterministic versions of models and progressing to stochastic (Monte Carlo) versions of models. The development of the economic-ecologic simulation model outlined in chapter 2 should begin first. The development of the mixed-integer programming model outlined in chapter 3 should proceed as soon as possible thereafter based on the information and data gathered for the development of the simulation model. After development of the economic-ecologic simulation model is well underway, consideration should be given to placing this model within a Monte Carlo simulation framework. The decision to place the mixed-integer linear programming model within a Monte Carlo simulation framework should be made after experience has been gained with the development and use of the economic-ecologic simulation model.

CONDITIONS ASSUMED FOR THE COST-EFFECTIVENESS ANALYSES

This section describes the biological objectives and other conditions that were assumed for the design and development of methods of cost-effectiveness analysis presented in this part. The biological objectives established for the anadromous fisheries, the economic criterion used to compare fish mitigation alternatives, the types of costs included in the cost-effectiveness analyses, and the nature of the policy constraints imposed on the analyses all influence the design and development of methods.

Biological Objectives

The biological objective adopted for purposes of model design and development is adult fish production, as measured by the number of adult fish harvested in the ocean and those that return to the Columbia River and are either harvested or spawn. There are other biological objectives that might have been adopted for this purpose, but the adult fish production objective is the most demanding from the perspective of model design and development. It requires a system-wide approach and it requires consideration of the full life-cycle of anadromous fish. Other biological objectives, such as the number of juvenile fish of a particular stock produced in a particular subbasin and the number of juvenile fish of a particular stock that arrive safely in the estuary below Bonneville Dam, are not as demanding in that they either do not require a system-wide approach to model design and development or they require consideration of only a portion of the fish life-cycle, or both. If a model can be designed and developed to satisfy the first biological objective, that same model will satisfy other, less demanding biological objectives.

The relationship between harvest rates in the ocean and harvest rates in the river in any given year is a management goal in the cost-effectiveness analyses described in this part. It is not an output of the analysis. The simulation model is not designed to explore economic tradeoffs between levels of ocean harvest and levels of in-river harvest. The relationship between these two conflicting management goals must be established exogeneously. However, it is important to recognize that ocean fisheries management bears importantly upon the availability of fish for upstream runs. Although the models discussed in this part do not incorporate ocean management as an option in the analysis, the models can accommodate as external information the types of economic information that would result from the ocean fishery studies proposed in Part III. This information would be needed for a biological objective that focused exclusively on the number of adult fish that escape the ocean and return to the Columbia River either to be harvested or to spawn. (This is a subtle point, but it is an important one if the cost-effectiveness analysis is to be done correctly.)

The relationship between harvest rates in any given year and the number of adult fish that are permitted to spawn also is an exogenous input to the simulation analysis and not an output of the analysis. (If a mathematical programming model of the anadromous fishery can be developed, the harvest rates could be an output of that model.) The relationship between the number harvested and the number allowed to spawn is not determined solely by biological considerations, but rather by the target levels established for the ocean and in-river harvests each year of the planning period (see discussion below), the levels and timing of investments in mitigation measures, and the biological relationships involving anadromous fish. Moreover, this relationship can change from year to year throughout the 20 year planning period, depending on natural and biological conditions and on the particular cost-effectiveness criterion adopted for the analysis.

Thus, the economic-ecologic simulation model outlined in chapter 2 is designed to meet target levels of adult fish for an exogeneously specified mix of ocean harvest, in-river harvest, and spawners. This multi-dimensional management goal may be expressed as fractions of the total available adults in particular target years, and alternative sets of goals may be specified. An illustrative example is shown in the following table.

Alternative Management Goals (Illustrative)	Fraction of Total Adults in a Particular Year			
	Ocean Harvest	In-River Harvest	Spawners	Total
A	0.4	0.4	0.2	1.0
B	0.6	0.2	0.2	1.0
C	0.2	0.6	0.2	1.0
D	0.3	0.3	0.4	1.0

The biological objective defined above as the total number of adult fish that are harvested in the ocean and that return to the Columbia River and are either harvested or spawn has been refined further for purposes of model design and development into stocks which are identified by species, spawning location, and natural (wild) or hatchery production. More than 30 stocks and areas of emphasis may eventually be identified by the Council for purposes of planning and management (Northwest Power Planning Council, 1987a and 1987b).

The biological objective also has been subdivided for purposes of model design and development into a sequence of time periods spanning one year each such that the biological objective (target level of total adult fish, distributed among the three categories of adults described above) can be specified over time. This recognizes that it will take a number of years for stocks to achieve target levels, whatever the goal. Moreover, there are alternative time paths of recovery that can be specified, each with different implications for the mitigation measures employed and the timing of those measures, and thus with different implications for costs.

Examples of two alternative time paths of recovery are shown schematically in Figure 1.2. The biological objective including the particular stocks and areas of emphasis to be considered, the distribution of ocean and in-river harvests, and the time path of recovery all must be specified for the cost-effectiveness analysis. These will ultimately result from planning processes currently underway in the region (Northwest Power Planning Council, 1987a and 1987b). In the interim, more or less arbitrary goals must be specified for purposes of model design and development.

Economic Criterion and Costs

The economic criterion used for model design and development in this part is cost-effectiveness. This has been defined operationally for purposes of model design and development to include three kinds of analyses: (1) an assessment and comparison of the economic costs of a set of alternative management strategies that achieve the same level of effectiveness (level of adult fish production); (2) an assessment and comparison of the economic costs and levels of adult fish production of a set of alternative management strategies that achieve different levels of effectiveness; and (3) identification of the most cost-effective management strategy for achieving a particular biological objective, subject to a set of administrative, legal, and political considerations imposed on the analysis.

Economic costs are defined as changes in total social costs due to implementation of a particular management strategy. In practice, costs are measured as changes in the sum of the direct and indirect costs (discussed below) due to the primary (but not secondary) effects of a set of protection, mitigation, or enhancement measures. Thus, costs are measured relative to

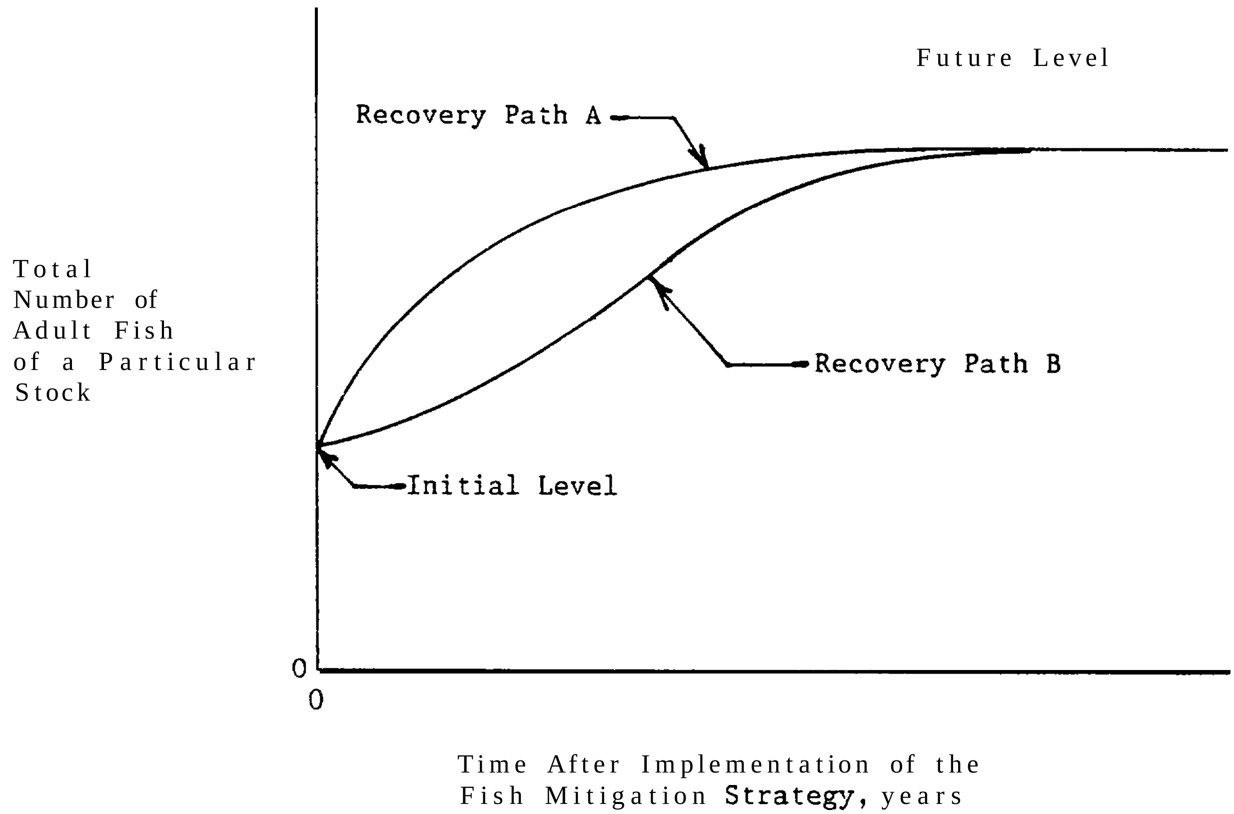

Figure 1.2. Alternative Time Paths of Recovery of a Particular Stock After Implementation of Fish Mitigation Strategy (Illustrative)

conditions that are projected to exist in the basin throughout the 20 year planning period, with and without the fishery management strategy in place.

The costs included in the analysis are those that are incurred in the U.S. and Canadian portions of the basin. (For some analyses, the opportunity costs of reductions in the ocean harvest also may be included, although as explained above measurement of those costs are reserved for discussion in Part III.) Changes in the operations of the storage projects in the Canadian portion of the basin are assumed to be part of the cost-effectiveness analysis, although the operations of these projects could be fixed at current levels if this alternative is not a viable one. In comparing the costs of alternative management strategies, costs that occur at different times in the analysis will be presented both as a stream over time and as discounted present values.

A distinction is made in this analysis between two types of costs--economic costs and financial costs. Economic costs are total social costs to whomever they may accrue. They are calculated as the sum of the public and private costs. Financial costs are private costs borne by particular economic activities. They are calculated as actual financial outlays for capital, labor, and land, net of subsidies and other income transfers. Economic costs are used in the cost-effectiveness analyses described in this part. Financial costs borne by particular sectors and activities in the basin are not evaluated in the proposed approach, although they could be at a later stage.

Two types of economic costs are considered in this analysis. The first type is the direct resource costs of fish mitigation measures. These include the capital, operating and maintenance, and land costs of fish hatcheries, equipment used to transport juvenile fish through and around dams, and adult fish ladders. Direct resource costs are relatively straightforward to estimate, assuming that market prices are available and that they can be used to estimate economic costs. Estimation of the direct resource costs of mitigation measures will be based on engineering cost estimation and on standard principles of engineering economy (Grant, Ireson, Leavenworth, 1982). Therefore, there is no need to elaborate on them further in this part.

The second type of economic cost is the indirect resource costs of mitigation measures. These costs, referred to by economists as "opportunity costs", are associated with changes in the operation of the hydrosystem for the benefit of anadromous fish. Two types of opportunity costs are proposed to be estimated and included in the cost-effectiveness analyses. They include the economic losses due to:

- reductions in the generation of hydropower for the benefit of anadromous fish (in both the U.S. and Canadian portions of the basin); and
- reductions in withdrawals of irrigation water through the purchase of water rights for the benefit of anadromous fish. (Measurement of the opportunity costs of reductions in withdrawals of irrigation water is discussed in detail in chapter 5 of this part.)

A third type of opportunity cost may be included in some analyses:

- regulation of the commercial and recreational ocean harvests of anadromous fish (for the benefit of increasing the in-river harvest or for increasing long-term production of anadromous fish, or both).

To elaborate a bit on this third type, if the level of the commercial ocean harvest is one of the management goals (targets) to be met (see the discussion above), it is not appropriate to include the opportunity cost of reductions in the ocean harvest in the cost-effectiveness analysis. If, on the other hand, the level of the commercial ocean harvest is not one of the management goals (targets) to be met, and if a reduction in the commercial ocean harvest is merely another way of increasing long-term levels of fish production, then the opportunity cost of reductions (as well as the gains due to subsequent increases) in the ocean harvest should be included in the cost-effectiveness analysis. In the latter analysis, the opportunity cost of reductions (and the gains due to subsequent increases) in the ocean harvest is compared with, and traded off against, the direct and indirect costs of all other mitigation measures considered in the analysis. (This is a subtle point, but one that needs to be taken into consideration in the design and development of models if the cost-effectiveness analysis is to be done properly.)

One other type of opportunity cost exists, but it will not be estimated or included in the cost-effectiveness analysis. This includes the economic losses due to changes in the operating rule curves at the storage reservoirs in the U.S. and Canadian portions of the basin that benefit anadromous fish at the expense of other beneficial uses of the reservoir such as flat-water recreation and flood control.

Opportunity costs are considerably more difficult to estimate than the direct resource costs, so they are discussed in more detail in chapter 5.

Policy Constraints

A variety of constraints are imposed on the cost-effectiveness analyses. The different types include technical, legal, institutional, political, and management (administrative) constraints. Some of these constraints can be changed in the short-run. Others can be modified over the long-run. Still others cannot be changed at all. The legal, institutional, political, and management constraints that are subject to negotiation and thus to change are referred to collectively as policy constraints.

The legal constraints include contracts with electric utilities both within and outside the region for firm hydroelectric power; treaties with Indian tribes in the basin concerning fishing rights and irrigation water rights; international treaties with Canada involving the storage and withdrawal of water in the Canadian portion of the basin and contracts for the delivery of electricity; irrigation water rights; and individual state laws.

Examples of management (administrative) constraints include the upper and lower rule curves at the storage reservoirs in the basin, and the requirement that BPA must provide funds to the states and to fish and wildlife agencies to operate fish hatcheries.

The constraints on the various cost-effectiveness analyses will need to be identified for the development of both the economic-ecologic simulation model (chapter 2) and the mathematical programming model (chapter 3). Both model structures are flexible in handling such constraints.

Planning Period

The planning period assumed for the analysis of fish production in the Columbia River Basin ranges from one season (several months) to 20 years, depending on the problem type (described in the next section). For a cost-effectiveness analysis of juvenile fish production, the planning period is one season. For a cost-effectiveness analysis of adult fish production, the planning period is 20 years. A planning period this long is necessary for adult fish production because it will take several fish life-cycles of between 4 and 6 years each to build up stocks to new levels. In addition, a planning period of 20 years is consistent with the planning period used by the Council to project future electrical energy needs in the Pacific Northwest (Northwest Power Planning Council, 1986a and 1987b).

PROBLEM TYPES

Different problem types can be identified depending on the biological objective established for the anadromous fisheries in the basin. Each objective implies a different geographic scope of analysis, a different set of biological components to be included in the analysis, different mitigation measures, different costs to be included in the analysis, and a different planning period for the analysis. Three problem types are described in this section, in order of increasing complexity, to illustrate the nature of the different problem types.

Juvenile Fish Production

The first problem type involves the production of a particular stock of juvenile fish. The biological objective for this problem type is the number of juvenile fish of a particular stock that is produced in a particular subbasin. The geographic scope of the analysis is the subbasin. The relevant portion of the salmonid life-cycle to be included in the analysis is from the laying and fertilization of eggs, though the rearing of fry, to the beginning of the smoltification process and the down-stream migration of smolts. The planning period for this problem type is one season.

Juvenile Fish Production and Migration

The second problem type is an extension of the first problem type. It involves the production of a particular stock of juvenile fish and their subsequent migration down the mainstem Columbia (or mainstem Snake) River to the estuary below Bonneville Dam. The biological objective for this problem type is the number of juvenile fish of a particular stock that reach the estuary. The geographic scope of the analysis is the tributary subbasin and the mainstem Columbia (or Snake) River to the estuary below Bonneville Dam. The relevant portion of the salmonid life-cycle to be included in the analysis is from the laying and fertilization of eggs, through the rearing of fry, to the migration of smolts to the estuary below Bonneville Dam. The planning period for this second problem type also is one season.

Adult Fish Production

The third problem type involves the production of a particular stock of adult fish. It comprises the entire salmonid life-cycle from the laying and fertilization of eggs, through the rearing of fry, the down-stream migration of juvenile fish to the ocean, and the ocean fishery, to the return of adult fish to the Columbia River and the arrival of spawners at hatcheries and natural spawning areas. The biological objective for this third problem type is the total number of adult fish harvested in the ocean and that return to the Columbia River to be harvested or to spawn. The geographic scope of the analysis is the portion of the Columbia River Basin that supports the salmon and steelhead fisheries, together with the ocean. The planning period for this third problem type is several fish life-cycles of between 4 and 6 years each. For purposes of model design and development, the planning period assumed for this third problem type is 20 years.

The characteristics and features of the three problem types described above are summarized in Table 1.1. As shown in this table, the three problem types have different characteristics and as such require different cost-effectiveness analyses. Moreover, the results of those analyses may imply different management strategies for the Columbia River Basin.

The biological objective established for the Columbia River fisheries completely specifies the problem type. Table 1.1 illustrates the importance of defining the biological objective before undertaking an analysis of the cost-effectiveness of alternative management strategies for the Columbia River Basin.

PROPOSED MODEL DEVELOPMENT

The development of an economic-ecologic simulation model of the Columbia River salmon and steelhead fisheries is proposed for the next phase of the research. This model would be capable of analyzing the cost and fish production implications of a set of prespecified management strategies. Experiments with the development of a mathematical programming model also is proposed. This model would be used to assist in identifying cost-effective management strategies for improving the Columbia River fisheries. An

Table 1.1. Taxonomy of Problem Types for Cost-Effectiveness Analyses of Fish Mitigation Strategies

Problem Type	Biological Objective	Geographic Scope of Analysis	Planning Period	Type of Fish Production Model	Biological Components in Analysis
Juvenile Production	Number of Juvenile Fish Produced	Subbasin	1 Season	Juvenile Production	0 Egg Production o Juvenile Rearing
Juvenile Production and Migration	Number of Juvenile Fish Reaching the Estuary	Subbasin plus Mainstem Columbia River	1 Season	Juvenile Production and Migration	0 Egg Production o Juvenile Rearing o Juvenile Migration
Adult Fish Production and Migration	Number of Adult Fish Harvested in the Ocean and Escaping the ocean	Columbia River Basin plus ocean	20 Years	Fish Life-cycle	0 Egg Production o Juvenile Rearing o Juvenile Migration 0 Early Ocean Survival 0 Ocean Survival 0 Adult Migration

illustrative application of these two models to a particular subbasin would test the potential usefulness of the models.

Both models, if successfully developed, will be used to analyze tradeoffs between levels of fish production and the costs of producing and maintaining those levels. Both models are needed since they fulfill fundamentally different purposes and have special strengths (and weaknesses). However, the effort to develop the mathematical programming model described in chapter 3 will be treading on entirely new ground, so it must be viewed as being more experimental than the simulation model at this stage.

OVERVIEW OF PART II

An overview of the principal elements and features of the proposed economic-ecologic simulation model is presented in chapter 2. The proposed mathematical programming model is described in considerable detail in chapter 3. The feasibility of applying that model to the design of cost-effective mitigation strategies for the Columbia River Basin is assessed in chapter 4. Approaches to estimating the opportunity costs of reductions in the generation of hydropower and of reductions in withdrawals of irrigation water for the benefit of anadromous fish are presented in chapter 5. The data that are needed to fully develop and to apply the two cost-effectiveness models presented in this part of the report are described in chapter 6.

The economic-ecologic simulation model in chapter 2 is not presented in as much detail as the fish production simulation models in Part I or the mixed-integer linear programming model in chapter 3 of this part. This is because the economic-ecologic simulation model lies more nearly within the existing state of the art and therefore less effort was expended on developing an appraisal of its feasibility in this phase of the research. Moreover, the economic-ecologic simulation model depends on the availability of a number of other component simulation models that either already exist in the region, such as the Systems Analysis Model (PNUCC, 1983) and the BPA hydrosystem regulator (BPA, 1984), or that are to be developed in other parts of the proposed research, such as the fish production simulation models described in Part I. However, this is not meant to imply that the development of the economic-ecologic simulation model will not require considerable effort. It will. Substantial amounts of data will be required, and the computer programming required both to develop and to integrate the component simulation models will be demanding.

Chapter 2

Assessing the Cost-Effectiveness of Alternative Mitigation Strategies: The Simulation Model

INTRODUCTION

This chapter describes the principal elements and features of an economic-ecologic simulation model designed to assess the costs and the long-term fish production implications of alternative management strategies for enhancing the salmon and steelhead fisheries in the Columbia River Basin. The design of the model is based on the availability of an energy planning model for the Pacific Northwest, an hydrosystem simulation model of the Columbia River Basin, and an hydrosystem regulator that already exist in the region, and on a simulation model of the salmon and steelhead fisheries to be developed in another part of this project (see description in Part I). It is important to note that such a model has not actually been developed for the Columbia Basin but that what is discussed here is an outline of how such a model could be developed. Actual construction of the quantitative model would be conducted in Phase III.

This chapter is organized in three principal sections. The first section is an overview of the economic-ecologic simulation model including brief descriptions of the three principal components of this model--a hydrosystem simulation model, a fish production simulation model, and a cost evaluation model. The description of the economic-ecologic simulation model in this section is a simplification of the model presented in more detail in the last section.

The second section is a review of some of the more promising electric energy and hydrosystem simulation models currently being used for planning and management in the Columbia River Basin. These include the Systems

Analysis Model used by BPA and others in the Pacific Northwest (PNUCC, 1983), the hydro regulators used by BPA (BPA, 1984), and the Hydrosystem Seasonal Regulation model used by the Corps of Engineers (Corps of Engineers, 1982). There also are two fish production and migration simulation models currently being used to simulate anadromous fish production in the Columbia River Basin. These include FISHPASS developed by the Corps of Engineers (Tanovan, 1985; Tanovan, Arndt, and Smith, 1987) and the Columbia River Basin Fishery Planning Model used for long-range planning by the Northwest Power Planning Council (Northwest Power Planning Council, 1986b). These two models are described in Part I of this report.

The third section describes the economic-ecologic simulation model proposed for development. This model is comprised of four principal parts--a master control module, a hydrosystem simulation model, a fish production simulation model, and a cost evaluation model. The hydrosystem simulation model simulates the monthly operations of the hydrosystem throughout the 20 year planning period assumed for the analysis. This model incorporates parts of the Systems Analysis Model and BPA's hydro regulator, and it includes those mitigation measures that pertain to the hydrosystem such as by-pass facilities at hydro projects to aid the passage of juvenile fish. The fish production simulation model simulates the production of salmon and steelhead trout throughout the 20 year planning period. This model comprises the biological portion of the economic-ecologic simulation model and includes those mitigation measures that pertain to the ecosystem such as improvements of natural spawning and natural rearing habitats. The fish production simulation model is described in considerable detail in Part I. The cost evaluation model is used to estimate both the direct costs of mitigation measures and the opportunity costs of measures. The master control module integrates the three principal components of the economic-ecologic model (hydrosystem, fish production, and cost evaluation models) and it controls the flow of information among these three components.

OVERVIEW OF ECONOMIC-ECOLOGIC SIMULATION MODEL

As mentioned above, the economic-ecologic simulation model is comprised of four principal parts--a master control module, a hydrosystem simulation model, a fish production simulation model, and a cost evaluation model. The relationship among these four parts is shown schematically in Figure 2.1. This schematic is a simplification of the actual model which is described in more detail in the last section of this chapter.

The master control module coordinates the flow of information to and from the other components of the economic-ecologic simulation model. Given a set of mitigation measures to be simulated, this module assigns values to the management variables (variables that are related directly to mitigation measures such as sizes of hatcheries) and it assigns initial levels to the state variables (variables that describe the state of the system such as streamflows and the number of adult spawners of a particular stock) in the hydrosystem simulation model and the fish production simulation model. This module also organizes and reports the results of the simulation analysis--the levels of adult fish production and the costs of producing those levels.

The hydrosystem simulation model mimics the unregulated flows in the tributaries to the Columbia and Snake rivers, the operations of the storage projects in the U.S. and Canadian portions of the basin, the operations of the run-of-the river projects, the flows and spills and the generation of electricity at hydro projects, the withdrawals of irrigation water, and the regulated flows in the Columbia River system to the estuary below Bonneville Dam.

The fish production simulation model mimics the production of juvenile fish and the migration of smolts to the estuary below Bonneville Dam, the ocean and in-river fisheries, and the migration of adult fish up the Columbia and Snake rivers and their tributaries to hatcheries and natural spawning areas.

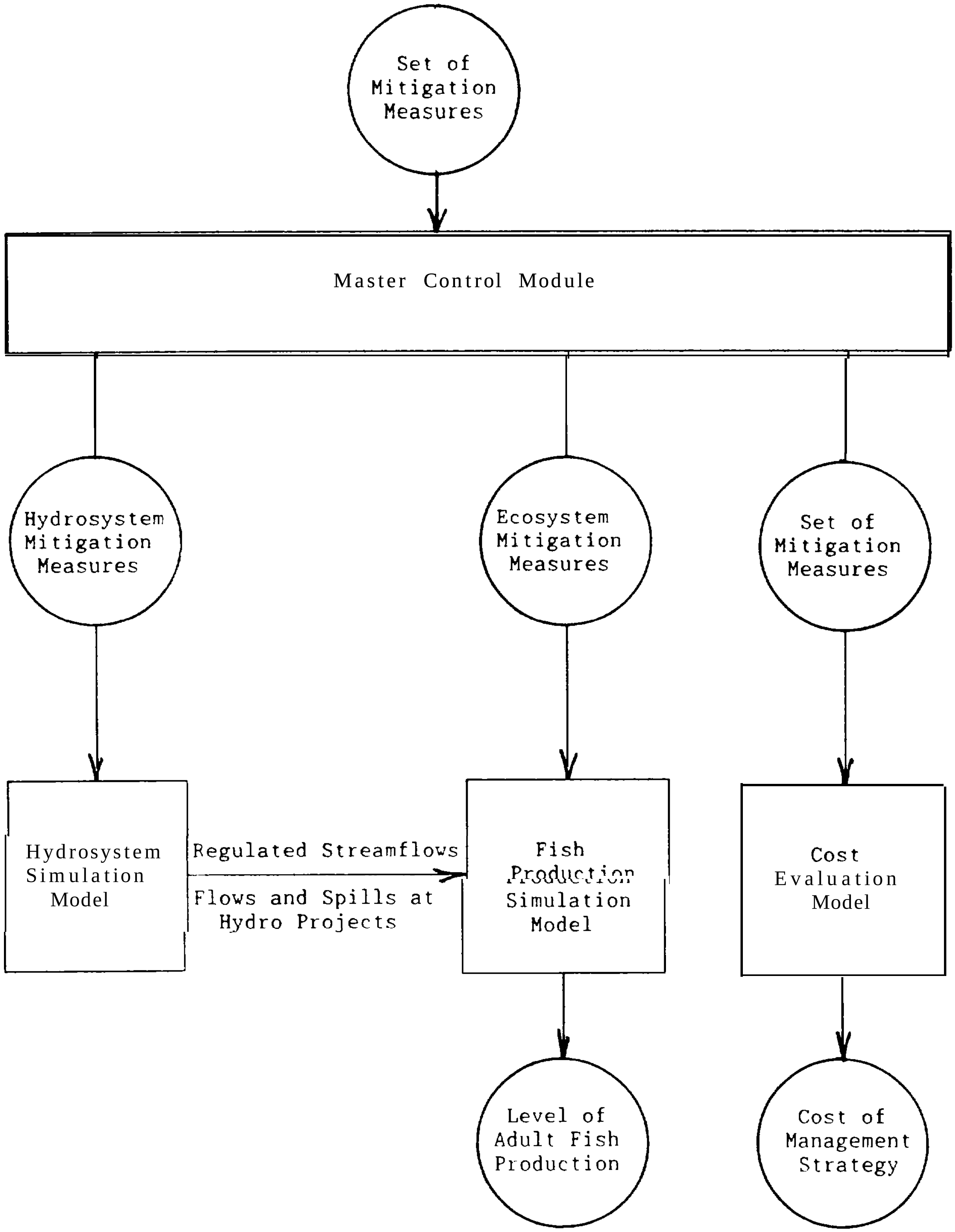

Figure 2.1 Schematic Diagram of Economic-Ecologic Simulation Model for Cost-Effectiveness Analyses

The cost evaluation model estimates both the direct resource costs and the opportunity costs of management strategies. It provides a time stream of future costs and it discounts future costs to present values to enable comparisons of alternative management strategies.

Hydrosystem Simulation Model

Hydrosystem simulation models are used widely in water resources planning and management. There are two basic types. One type is called a mass-balance model. For this type, the time step of the simulation cannot be shorter than the time it takes for a release of water from an upstream reservoir to be noticed at the mouth of the river. A typical time step for the mass-balance type simulation model is one month. Thus, streamflows in this type of simulation are typically analyzed and reported as monthly streamflows, and all the management variables and all other state variables in the simulation are reported on the basis of monthly averages. This type of model is used most often for the design and long-term operations of storage reservoirs.

The other type of hydrosystem simulation model is called an hydraulic routing model. For this type, the time step of the simulation can be as short as it needs to be to describe the particular phenomenon. A typical time step for flood routing is one hour to several hours. The hydraulic routing simulation model is considerably more difficult to build than the mass-balance simulation model, and it requires considerably more data. The hydraulic routing type hydrosystem simulation model is used to assess the nature and extent of potential future floods and to design flood control levees. This type of model may ultimately be required to assess the impacts of hydrosystem operations on anadromous fish.

A mass-balance type hydrosystem simulation model is proposed for the next phase of the research. This decision is based on two principal considerations. First, the economic-ecologic simulation model is designed to assist in long-term strategic planning. It is not designed to support short-term management or operational decisions. A monthly time step for the hydrosystem simulation seems sufficient to assess long-term impacts on the salmon and steelhead fisheries. Second, a mass-balance hydrosystem model is considerably easier to build, and certainly less expensive, though still consumptive of a large effort, than an hydraulic routing hydrosystem model.

The time-step in the proposed hydrosystem simulation model is one month. Average monthly conditions, and conditions at the beginning and end of each month, are used to describe the essential features of the hydrosystem such as levels of the storage reservoirs, streamflows, water withdrawals for irrigation, and the production of hydroelectric power, and average monthly conditions are used to estimate mitigation costs and to assess impacts on anadromous fish. Short-term episodic conditions lasting less than one month are not reflected in this model. If short-term episodic conditions such as flooding and flood control and releases for peak power demands turn out to be important for long-term strategic planning, the short-term phenomena must either be related a priori to state variables in the hydrosystem simulation model measured as monthly averages or they must be simulated using a specially designed hydraulic routing type simulation model or set of simulation models with a time step shorter than one month.

A short-term hydraulic routing model is not described in this report and it is not proposed for the next phase of the research. The reason for this is that it is not known at this stage in the research if short-term phenomena will require special attention in order to inform long-term strategic planning, and it will not be known until after the economic-ecologic simulation model has been constructed and analyses using that model have been made. If short-term phenomena do require special attention and analysis, it will be considered experimental and it will require an extension of the methods proposed in this part of the report.

An hydrosystem regulator, driven by the demand for electrical energy in the region and by the availability of water in the basin, will be used in conjunction with the hydrosystem simulation model (described above) to allocate the generation of hydroelectric power among the hydro projects in the basin. This, in turn, defines the monthly operations of hydro projects in terms of flows and spills, within the rule curves established for other uses of the reservoir such as flood control and flat water recreation. Once monthly flows and spills at the hydro projects in the basin have been determined, the hydrosystem model will be used to account for the regulated monthly flows in the river system and for the levels of reservoirs at the storage projects and the levels of pools at the run-of-the-river dams.

The principal inputs to the hydrosystem simulation model are the unregulated monthly streamflows, the initial reservoir levels at the storage projects and initial pool levels at the run-of-the-river dams, and the levels of the hydrosystem management variables such as irrigation water withdrawals and flows and spills at hydro projects, including the proportions of total flows (at hydro projects) that pass through the turbines, through fish by-pass conduits, and over spillways. (These proportions are determined partly by considerations for the survival of juvenile fish passing hydro projects and partly by the hydrosystem regulator which was described briefly above and in more detail in the last principal section of this chapter.) Some of these inputs are provided by the master control module for each management strategy analyzed (Figure 2.1). Other inputs are provided by the hydrosystem regulator.

There are several sets of reservoir rule curves that must be developed prior to hydrosystem simulation, one set for each of the storage projects in the basin. These rule curves are used to guide the monthly operations of the storage projects. They are based on a number of factors and considerations including projections of energy requirements (loads), historic streamflows, water budgets for the passage of juvenile fish, and flood control. There are several different kinds of rule curves, some based on flood protection and others based on firm power production. An operating rule curve based on all the other rule curves is used for normal operations of the storage projects in the basin. (Development of reservoir rule curves is discussed in more detail below.)

The principal outputs of the hydrosystem simulation model are the monthly regulated streamflows in the basin and the average time of passage of water through the reservoirs, by month and year. These outputs together with the flows at the hydro projects that pass through the turbines and the spills that pass through fish by-pass facilities and over spillways are passed along to the fish production simulation model.

Fish Production Simulation Model

Fish production simulation models typically are not as well developed as hydrosystem simulation models. This is due partly to the more complicated biological relationships in fish production models, partly to the lack of biological data, and partly to the more recent use of mathematical models and computers in the analysis of ecosystems. As previously noted, an approach to modeling fish production in the Columbia River system has been developed in another part of this project. The fish production models described in Part I of this report will be used in the proposed economic-ecologic simulation model. They are not discussed further in this part except to describe the linkages between the fish production simulation model and other parts of the economic-ecologic simulation model (Figure 2.1).

There are two groups of inputs to the fish production simulation model. One group is provided by the master control module. The other group is provided by the hydrosystem simulation model. The principal inputs from the master control module are the levels of the management variables that pertain to the ocean and in-river fisheries such as the number of fry or smolts that are released from hatcheries, the capacities of natural spawning and natural rearing habitats, and the distribution of the ocean harvest, the in-river harvest, and the adult fish that are permitted to spawn. The principal inputs from the hydrosystem simulation model include the monthly regulated streamflows throughout the basin, the average time of passage of water through the reservoirs by month of year, and the monthly flows and spills at hydro projects that pass through the turbines, through fish by-pass facilities, and over the spillways.

The principal outputs of the fish production simulation model are the levels of adult fish production measured by the number of adult fish that are harvested in the ocean, the number that are harvested in the river, and the number that are permitted to spawn, for each year in the planning period and for each stock considered in the analysis.

Cost Evaluation Model

The cost evaluation model estimates both the direct resource costs and the indirect resource (opportunity) costs of each management strategy analyzed, organizes these costs by the year they are incurred, and computes the present value of the time stream of costs.

REVIEW OF HYDROELECTRIC AND HYDROSYSTEM SIMULATION MODELS

This section contains brief summaries of electric energy and hydrosystem simulation models currently being used for planning and management in the Columbia River Basin that seem promising, in whole or in part, for incorporation in the proposed economic-ecologic simulation model. These include the Systems Analysis Model prepared by the Pacific Northwest Utilities Conference Committee (PNUCC, 1983), the hydro regulators developed and used by BPA (BPA, 1984), and the Hydrosystem Seasonal Regulation Model developed and used by the U.S. Army Corps of Engineers (Corps of Engineers, 1982). Brief descriptions of FISHPASS developed by the Corps of Engineers (Tanovan, 1985; Tanovan, Arndt, and Smith, 1987) and the Columbia River Basin Fishery Planning Model used for long-range planning by the Northwest Power Planning Council (Northwest Power Planning Council, 1986b) may be found in Part I of this report.

Systems Analysis Model (SAM)

The Systems Analysis Model (SAM) is a large, complex simulation model that serves many purposes within BPA, varying from long-range projections of thermal power generation requirements to analysis of the impacts of intertie expansion on hourly hydrosystem flows during the spring smolt migration. The economic-ecologic simulation model will require only a modest subset of SAM's overall capabilities, and this description concentrates on the subset we plan to incorporate in the economic-ecologic simulation model. The reader who is unfamiliar with SAM may wish to read the overview report on SAM (PNUCC, 1983).

SAM is a long-range, Monte Carlo, simulation model of electric energy supply and demand in the Pacific Northwest with a typical planning horizon of twenty years. It operates the hydroelectric portion of the system as though it were owned by a single firm, performing a probabilistic simulation of the region's power demands and power supplies. The model simulates both long-term planning operations, mimicking the annual and seasonal planning process, and short-term systems operations at intervals ranging from monthly to daily, and even hourly, operations. The proposed economic-ecologic simulation model will use both the planning mode and the monthly operating mode of SAM.

Of the many possible outputs from SAM, the needs of the proposed economic-ecologic simulation model focus on two broad areas. The first is the demand for electricity, disaggregated by firm power, direct service industry (DSI) loads, and sales outside the region. The second is the supply of electricity, disaggregated by thermal power and hydropower. SAM is also capable of producing information on the allocation of loads among individual hydro projects and thermal power generating plants, although it will be necessary to modify SAM in order to produce more accurate estimates of the monthly loads at individual hydro projects. This refinement is needed for simulating the impacts on fish of alternative mitigation strategies.

More specifically, we plan to use the output from a number of computer runs of SAM to generate loads, disaggregated into the categories noted above, and total generation, disaggregated into total thermal generation and total hydropower generation. The hydrosystem loads will be translated into individual operating rule curves at each of the hydro projects using one of BPA's hydro regulators (described briefly above and in more detail below).

In order to assess the trade-offs between improvements in adult fish production and reductions in hydropower generation, we plan to use SAM under a variety of flow scenarios. While the details of the scenarios have not yet been decided, we tentatively plan to alter flow and spill regimes systematically, thereby varying the amounts of hydropower that can be generated. This will be done outside the economic-ecologic simulation model. The operating rule curves that are output from SAM and the BPA hydro regulator will be used as inputs to the hydrosystem simulation model, as mentioned above and described in more detail below.

Hydro Regulator

Hydropower production in the Pacific Northwest is designed to meet firm energy loads. Therefore, determination of the firm load capability of the system is a critical part of the hydrosystem analysis. This capability is referred to in the Columbia context as the firm energy load carrying capability (FELCC) of the hydroelectric system. It is defined in the Pacific Northwest Coordination Agreement (BPA, 1964) as the maximum amount of energy, in the same monthly distribution as the system's firm energy loads, which the system is able to produce without failure throughout the historic period used for hydroelectric resource planning, using all of its reservoir storage in combination with its historic streamflows (Dean, 1982). The historic streamflows currently used for this analysis are the unregulated flows for the 40-year period, 1928-29 through 1967-68.

The accounting of flows, loads, and storage is commonly referred to as hydroelectric system regulation. For hydroelectric systems as complex as those in the Pacific Northwest, it is necessary to iterate using computer-prepared hydroelectric regulations to determine the system's FELCC and to identify the critical period (the period in the historic record where the usable reservoir storage is drafted from full to empty to produce the system's FELCC).

Hydro regulators are used by BPA for several purposes concerning the operation of the Columbia River hydrosystem (BPA, 1984). Two such purposes relevant to the proposed economic-ecologic simulation model are the determination of the firm energy load carrying capability (FELCC) of the hydroelectric system and the preparation of operating rule curves for the storage projects in the basin. The input data needed for hydro regulation analysis consists of historic streamflows, adjusted for modern depletions and hydroelectric development, and basic energy loads and resources, which may be historic but more often are forecasts from the present up to 20 or 30 years in the future (BPA, 1984). Forecasts of energy loads and resources are provided by the System Analysis Model discussed above.

One of the important outputs of the hydro regulator with respect to the proposed economic-ecologic simulation model is the set of operating rule curves for the storage projects in the basin. The operating rule curve which is used to guide the monthly operations of the reservoir is developed from four different rule curves: an upper rule curve which provides flood protection, an energy content curve, a critical rule curve, and a limiting rule curve. The last three rule curves pertain to the ability of the system to generate firm hydropower.

Other hydrosystem regulators also are available in the Pacific Northwest, but BPA's hydro regulator appears adequate to provide the information on hydro regulation that will be required as input to the economic-ecologic simulation model.

Hydrosystem Seasonal Regulation Model

The Hydrosystem Seasonal Regulation model (HYSSR) was developed by the Corps of Engineers and is used by both the Corps and other groups in the Pacific Northwest for a number of purposes (Corps of Engineers, 1982). For example, it is used by the Corps to provide input flows to FISHPASS (Tanovan, 1985). While the model can be used for hydrosystem regulation, and while it also can be used to simulate regulated streamflows, hydropower production, and flood control, we do not plan to use it for the proposed economic-ecologic simulation model since, as just discussed, we plan to use BPA's hydro regulator.

PROPOSED ECONOMIC-ECOLOGIC SIMULATION MODEL

The proposed economic-ecologic simulation model can now be described in a summary and integrated way. The model design is based on the information needs of BPA, the Northwest Power Planning Council, and other concerned parties in the Pacific Northwest, as described in chapter 1.

Features of the Simulation Model

As indicated previously, the economic-ecologic simulation model is intended to be used for long-term strategic planning for the salmon and steelhead fisheries in the Columbia River Basin. The model is designed to explore the cost and long-term fish production implications of alternative management strategies, including changes in the water budget used to aid the passage of juvenile fish. The principal outputs of the model are the long-term production of adult fish and the costs of producing and maintaining those levels.

There are several mitigation measures to be considered. They include additional hatchery capacity, improvements in natural spawning and natural rearing habitats, increased flows in rivers and increased flows over spillways during critical periods to aid the passage of juvenile fish, additional by-pass facilities at major hydro projects along the Columbia and Snake rivers, and transportation of smolts by truck and barge to the estuary below Bonneville Dam. These measures and others will be included in the proposed model.

The costs in the model include both the direct resource costs of fish mitigation measures and the opportunity costs of reductions (1) in the generation of hydropower and (2) in withdrawals of water for irrigation through the purchase of water rights, for the benefit of anadromous fish. Estimation of the opportunity costs of reductions in hydropower generation and of reductions in irrigation water withdrawals represents a major analytical undertaking. This is discussed in considerable detail in chapter 5.

The time step of the simulation will be one month. The length of the simulation will be for a period representing 20 or more years, depending upon the length of time it takes for anadromous fish populations to reach new target levels.

Elements of the Simulation Model

The principal elements of the economic-ecologic simulation model are described in this section. They include the fish mitigation measures to be incorporated in the simulation model and the components of the simulation model that will be used in a typical simulation run of the model.

Mitigation Measures. All applicable fish mitigation measures will either be incorporated directly in the economic-ecologic simulation model or simulated as part of the analysis, or both. Many of these measures were discussed previously in Part I and earlier in this part. Therefore, they will be mentioned only briefly here. For purposes of discussion, the measures are organized in four groups according to their effects on the life-cycle of anadromous fish. These four groups include: smolt production, smolt migration, estuarine and ocean survival, and upriver migration. General information on applicable mitigation measures are provided in the report of the Columbia River Basin Fish and Wildlife Program (Northwest Power Planning Council, 1987a). Details on particular mitigation measures are expected to be available from the Corps of Engineers and from subbasin planning studies currently underway (Northwest Power Planning Council, 1987a, 1987b).

Mitigation measures used to improve smolt production include additional hatchery capacity, improvements in natural spawning habitat, improvements in natural rearing habitat, outplanting policy, and modification of smoltification schedules.

Mitigation measures used to improve downstream migration of smolts include increased river flows, increased spills at hydro projects, fish by-pass facilities at hydro projects, and transportation of juvenile fish by truck and barge to the estuary below Bonneville Dam.

Mitigation measures used to improve upstream migration of adult fish include increased river flows, more efficient fish ladders, and increased flows over fish ladders.

Components of Model Used in Simulations. As previously described, the proposed economic-ecologic simulation model is comprised of four principal parts--a master control module, a hydrosystem simulation model, a fish production simulation model, and a cost evaluation model. The relationship among these four parts was shown previously in Figure 2.1.

Before the economic-ecologic model can be used for simulation, several inputs will be required from other models, for example, the Systems Analysis Model and the hydro regulator described previously. The purpose of this section is to describe the operation of the economic-ecologic simulation model, and the flow of information both from outside and within the model.

The operation of the proposed economic-ecologic simulation model is shown schematically in Figure 2.2. The two principal “driving forces” behind this operation are the projected energy requirements (loads) for the Pacific Northwest and the historic (unregulated) streamflows. Given a water budget for anadromous fish at Priest Rapids and Lower Granite Dams together with requirements for spills at hydro projects to aid the passage of juvenile fish, these two “driving forces” determine the firm hydropower available from the hydroelectric system and in turn are instrumental in the development of the operating rule curves for the storage projects in the U.S. portion of the Columbia River Basin.

Except for the development of the operating rule curves which is done outside the economic-ecologic simulation model, the model operates in a sequential fashion. The components of the model are discussed in the order they would appear in a typical simulation run of the model.

The economic-ecologic simulation model requires a set of rule curves for each of the storage projects in the basin before the model can be used for simulation. The upper rule curve for flood protection and the lower rule curve that defines the boundary between live storage and dead storage are fixed. These rule curves are available for each storage project from the Corps of Engineers (Corps of Engineers, 1985). However, the operating rule curves, which are used to guide the monthly operations of storage projects are not fixed and may have to be developed for each simulation run of the model that involves a change in the flow regime, depending on the conditions assumed for the analysis. The Systems Analysis Model and the hydro regulator shown in the upper portion of Figure 2.2 will be used to develop the rule curves needed for the economic-ecologic simulation model.

As mentioned previously, the Systems Analysis Model (SAM) projects future energy requirements (loads) for the Pacific Northwest and, in conjunction with the hydro regulator, generates thermal loads and resources and firm hydroelectric loads. The hydro regulator uses historic (unregulated) streamflows and, in conjunction with SAM, generates a set of energy rule curves for all the storage projects. These include the critical rule curve (CRC), the assured refill curve (ARC), the variable refill curve (VRC), and the lower limits energy content curve (LLECC). The energy rule curves, together with the mandatory (upper level) rule curve (MRC) for flood protection and the lower rule curve that defines the limits of active storage, are used to develop the operating rule curve (ORC). An example of the procedure used to develop an operating rule curve for a typical storage project is shown in Figure 2.3 (Columbia River Water Management Group, 1986).

The rule curves for the storage projects are one of the principal inputs to the hydrosystem simulation model. Other inputs include historic or

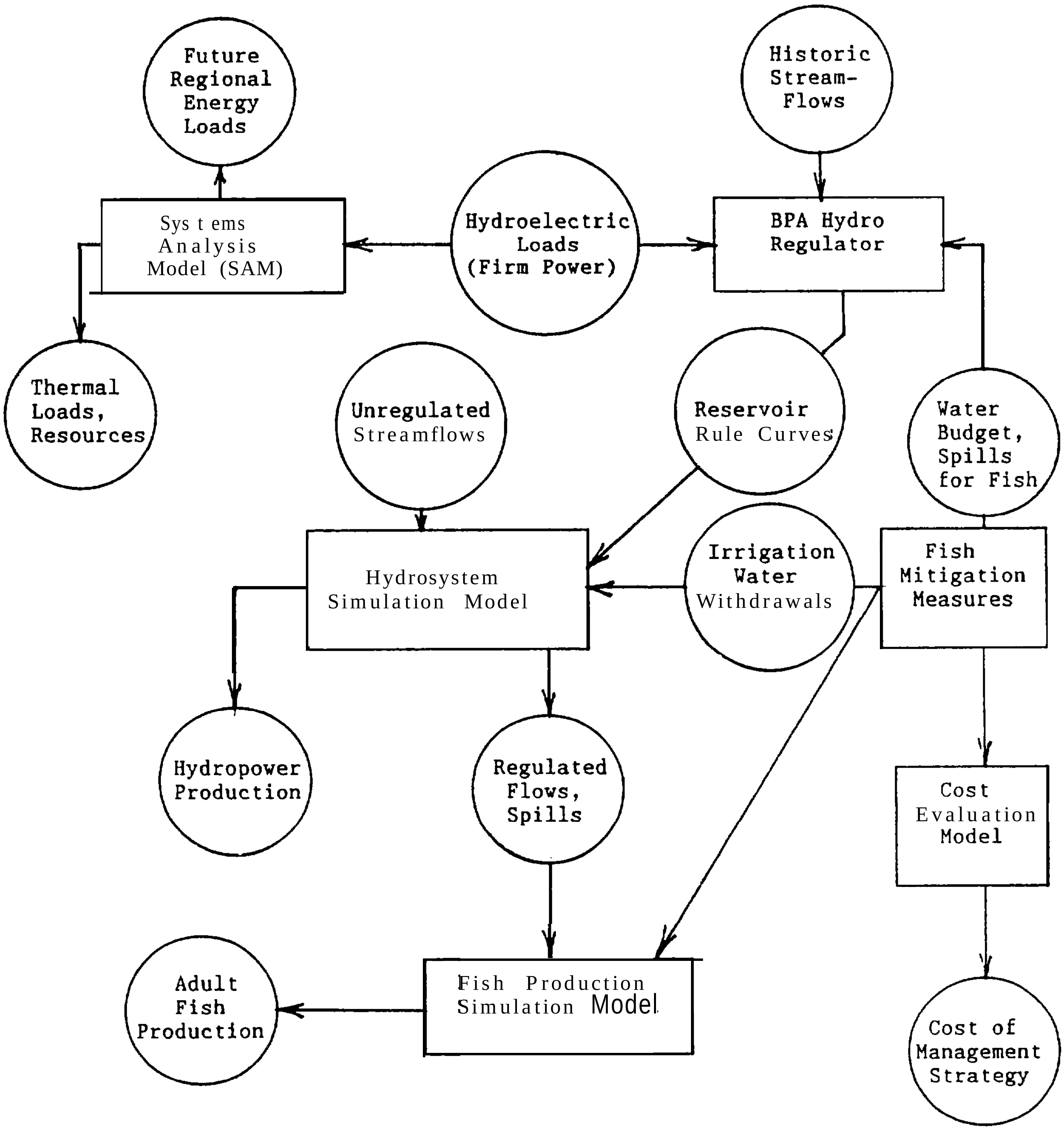

Figure 2.2. Schematic Diagram of the Operation of the Proposed Economic-Ecologic Simulation

RULE CURVE DEFINITIONS

MRC *MANDATORY RULE CURVE* **For the period August thru December is based on historical flows and for the period January thru July is based on forecast flows. The MRC's reflect the amount of storage space needed to protect against flood.**

CRC *CRITICAL RULE CURVE* **Is actually a family of one to four curves depending on the length of the critical period. These curves are** developed in July **of each operating year from historical flows and based on operating under adverse flow conditions.**

ARC *ASSURED REFILL CURVE* **This curve is theelevation that each project can refill if thesecond lowest historical wateryear(l931). January thru July run-off should occur.**

VRC *VARIABLE REFILL CURVE* **This curve depicts the reservoir elevation needed to refill with 95 percent assurance based on the current run-off forecast.**

ORC *OPERATING RULE CURVE*

(August thru December) The ORC is the higher of the ARC or the CRC unless the MRC is lower, then it controls.

(Januarythru March)TheORCmethod isthesameasthe August thru December period unless the VRC is lower, then it controls. When the VRC controls the ORC can be higher than the MRC. **But in no case can the** ORC **be lower than the** LLECC

(April thru July) The ORC **method is the same as January thru March period, except without the LLECC consideration.**

LLECC *LOWER LIMITS ENERGY CONTENT CURVE* **Protects the ability to meet Firm Load from forecast error.**

TYPICAL STORAGE PROJECT RULE CURVE OPERATION

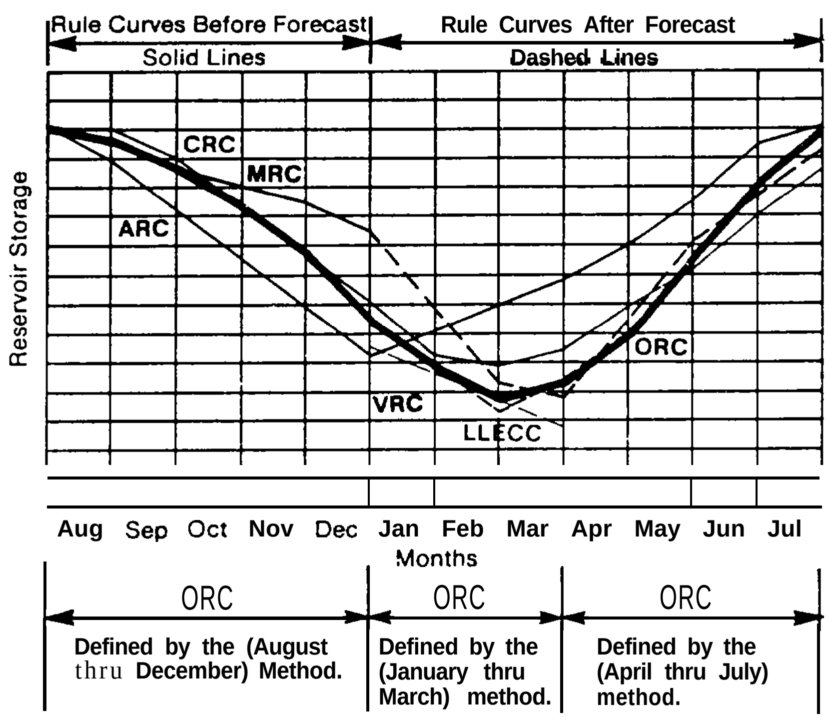

Figure 2.3. Procedure used to **Develop** the Operating Rule Curve for a Typical Storage Reservoir

Source: Columbia River Water Management Group. 1986. Columbia River Water Management Report for Water Year 1986, Columbia River Water Management Group, U.S. Army Corps of Engineers, North Pacific Division, Portland, Oregon, April.

generated streamflows; irrigation water withdrawals; the proportion of total flows passing through the turbines, fish by-pass facilities, and over spillways; minimum flows for fish (water budget); and the effects of other fish mitigation measures on the operations of the hydrosystem.

The principal outputs of the hydrosystem simulation model are regulated monthly streamflows, monthly flows through turbines and monthly spills through by-pass facilities and over spillways, and the generation of hydropower at the different hydro projects, disaggregated by firm (or primary) power and surplus (or secondary) power.

Information on streamflows from the hydrosystem simulation model is passed along to the fish production simulation model. This model together with information on the particular mitigation measures used in the simulation is used to project levels of adult fish of particular species and stocks. These projections of adult fish are used in the cost-effectiveness analyses described in chapter 1.

Information on the mitigation measures assumed for particular simulations, including reductions in the generation of hydropower and reductions in withdrawals of irrigation water for the benefit of anadromous fish, is passed along to the cost evaluation model where the costs incurred in particular years are estimated and where the present values of the time streams of those costs are computed. Both the direct resource costs and the opportunity costs of fish mitigation are included in the costs estimated in the cost evaluation model.

The mitigation costs and the levels of adult fish production for each management strategy analyzed are used to assess the cost-effectiveness of particular management strategies, as described in chapter 1.

CONCLUDING COMMENTS

The economic-ecologic simulation model described in this chapter is a deterministic model. After this model has been developed, it should be placed within a Monte Carlo simulation framework to provide a capability for assessing the levels of uncertainty in projections of fish production, mitigation costs, and measures of cost-effectiveness.

Both the deterministic and stochastic versions of the economic-ecologic simulation model will be provided with interactive computer software and they will be designed to be “user friendly”. The intent is to transfer models, data sets, and computer programs to BPA, and/or other interested and responsible parties, after completion of the research to enable their personnel to explore the fish production and cost implications of alternative management strategies for the Columbia River Basin and to examine the implications of different “what if" scenarios.

June 16 1988

Chapter 3

Identifying the Most Cost-Effective Fish Mitigation Strategy: The Least-Cost Model

INTRODUCTION

As indicated in both chapters 1 and 2, simulation is a flexible and powerful methodological tool for analyzing the cost and fish production implications of a prespecified set of fish mitigation alternatives. But when confronted with a large array of alternative mitigation options, not all of which can be implemented, the cost-effectiveness analysis may take the form of searching for the least-cost mitigation strategy that can achieve the biological objective. Using simulation in a trial-and-error approach to approximate a least-cost solution probably is a poor choice for searching.

Identifying the least-cost strategy through a systematic search procedure such as mathematical programming may be more efficient but is a much more difficult problem than assessing the hydrologic, ecologic, and economic implications of particular mitigation alternatives or experimenting with an exogenously specified set of mitigation alternatives using simulation. The analytical approaches to these two types of analyses are entirely different, and different skills, experiences, and disciplines are required for each. The former requires all the skills and experiences of the latter, and more.

The graph presented in Figure 3.1 illustrates the essence of the least-cost problem. (It also illustrates the essence of the cost-effectiveness problem more generally.) This graph shows the relationship between alternative levels of adult fish production for a particular species (or stock) and the costs of producing those levels. This graph is illustrative because, in general, the costs of mitigation cannot be estimated for an individual species or stock. Investments in mitigation procedures usually

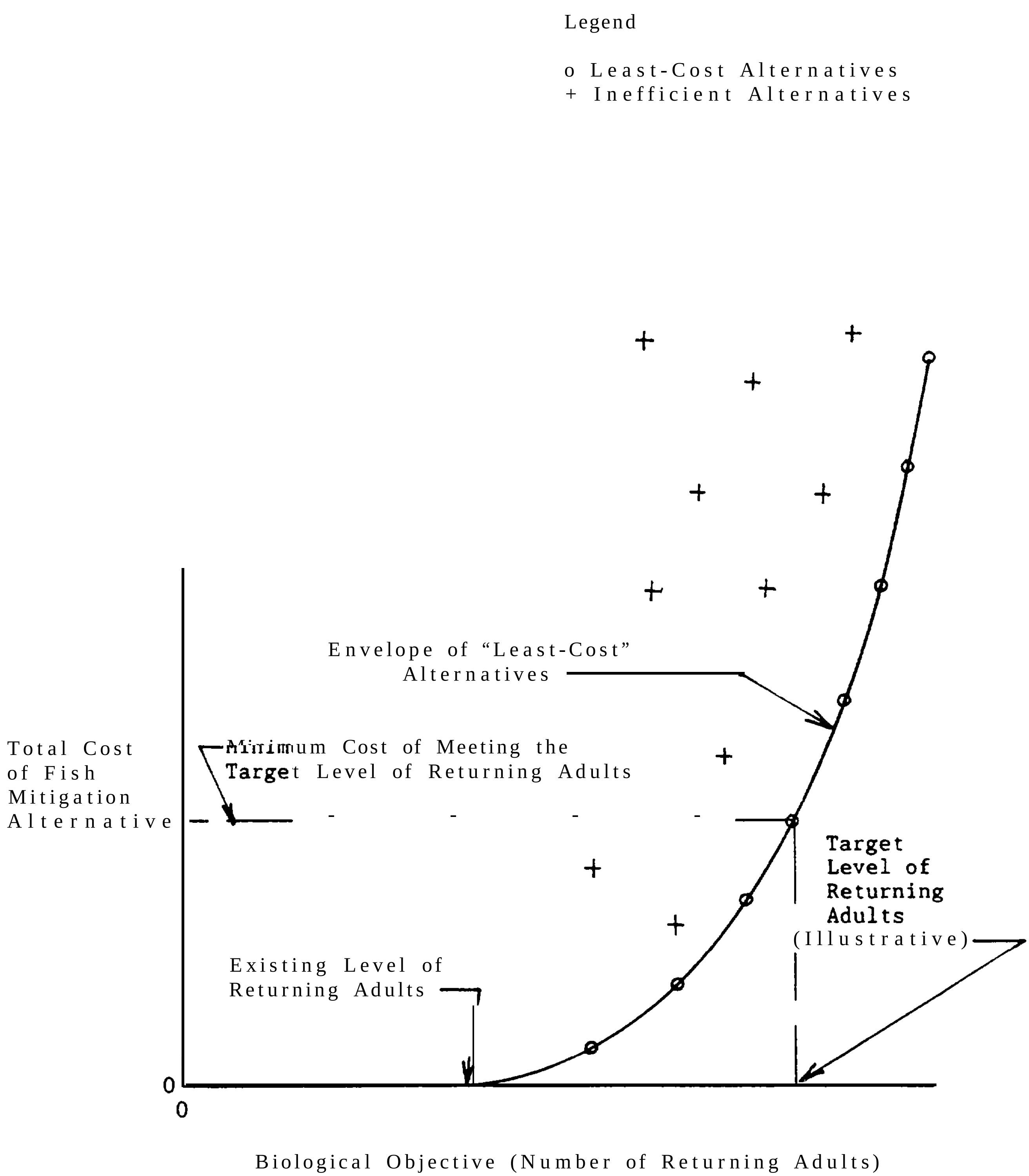

Figure 3.1. The Least-Cost Fish Mitigation Alternatives for a Particular Stock and for Different Biological Objectives

benefit several stocks and species simultaneously, and there is no unambiguous way of assigning joint costs to individual stocks or species.

The curve shown in this figure represents the “envelope” of system-wide least-cost fish mitigation alternatives for different target levels of returning adult fish. One of the principal strengths of mathematical programming is its ability to identify the alternatives that fall on this “envelope”. Also shown in this figure is a set of less cost-effective alternatives. These alternatives are indicated by the symbol "+" and lie above and to the left of the least-cost curve. Simulations of pre-specified fish mitigation alternatives typically produce such results.

The graph in Figure 3.1 illustrates the problem of attempting to identify the least-cost set of alternatives using a model designed for simulation rather than a model designed from the outset to identify the least-cost alternatives. Simulation models, while very useful and relatively easy to understand, are, by themselves, highly unlikely to identify the least-cost alternatives. This is because in a system with many possible control options the number of possible alternatives is astronomical and judgment and intuition are unlikely to lead to least-cost solutions. This is why it is necessary to try to develop a systematic search procedure even though this involves confronting substantial mathematical complexities.

Formal search procedures in this context take the form of mathematical programming models (linear programming, mixed-integer linear programming, and nonlinear programming). In this approach, the particular trial alternatives to be assessed are not specified a priori. Rather, the mathematical programming model is used to identify that combination of potentially available fish hatcheries, levels of enhancement of natural spawning and rearing areas, operations of dams and reservoirs, streamflows, and other mitigation alternatives that can achieve the biological objective at least-cost, subject to limits, such as electric power production requirements, called constraints. The set of least-cost results shown in Figure 3.1 is representative of the results which can in principle be obtained from mathematical programming models.

On the other hand, the simulation approach is relatively straightforward from a computational perspective. For the most part, this approach involves integrating various simulation models (e.g., cost, hydrosystem, and fish migration simulation models) and applying standard principles of engineering economy, as explained in chapter 1. The most difficult part of this approach is specifying the scenarios to be analyzed in searching for the least-cost alternative or set of alternatives. But we note again that simulation, because of its flexibility in mimicking the actual system, is the method of choice for projecting the outcomes of particular mitigation alternatives.

In addition to differences in the computational complexity of the mathematical programming and simulation approaches, there may also be differences in the accuracy of their outputs. Because mathematical programming models tend to grow large in size (measured by the number of management and state variables and by the number of constraining relationships) and thus to become difficult to manage and in some cases “unsolvable”, simplifying assumptions are often required. These simplifying assumptions can affect the accuracy of the results obtained from these models.

Assumptions made to reduce the size of mathematical programming models are generally not required of simulation models. Thus, simulation models are able to provide more accurate assessments of the costs and ecological implications of particular mitigation alternatives than programming models. Therein lies the dilemma for the least-cost analysis. Simulation models are able to provide more accurate assessments of the cost and the fish production implications of particular mitigation alternatives, but they are weak in their ability to identify the set of least-cost alternatives. Mathematical programming models are designed to identify the set of least-cost alternatives, but due to the simplifying assumptions that are generally required, they do not mimic reality with the fidelity of simulation models. Although each approach has desirable features, neither is entirely satisfactory for the needs of the Columbia River Basin fish mitigation analysis. A combination of the two approaches may be workable.

Perhaps the best way to view mathematical programming for large, complex economic-ecologic systems such as fish production in the Columbia River Basin is as a "screening" device to assist in identifying a set of technically feasible management alternatives that achieve the same level of effectiveness or that have desirable cost-effectiveness properties, or both. This set of alternatives can then be simulated using the more detailed economic-ecologic simulation model. The latter will provide more accurate estimates of the mitigation costs and more accurate estimates of the fish production implications of the various mitigation alternatives. These more accurate estimates of costs and of the numbers of adults available for harvest and spawning can then be used to identify the least-cost set of mitigation alternatives (see Figure 3.1).

Choice of a Model

The goal of this chapter is to identify the structure of a least-cost mathematical programming model and to estimate its size. Structure and size are the two principal attributes of such models that influence their success in real applications. For large scale applications such as this, the most desirable model structure from a computational perspective is a linear programming (LP) model. In order to be able to use linear programming, though, all the cost functions and all the constraining relationships must either be linear or be approximated by linear segments.

In the typical least-cost problem, it is not always possible to approximate all the nonlinear relationships with linear segments. Therefore, it is not always possible to use linear programming. However, it is sometimes possible to use integer variables to approximate the remaining nonlinear relationships. This requires the use of mixed integer linear programming, the next most desirable model structure for large scale applications. In this case, though, the number of integer variables becomes a practical consideration along with size (characterized by the total number of variables and the total number of constraining relationships) in real applications. If mixed integer linear programming cannot be used, it may be necessary to use nonlinear programming. This is the least desirable model structure for large scale applications. Nonlinear programming models typically consume large amounts of computer time and they do not guarantee identifying the “global” (as opposed to “local”) least-cost strategy, if the nonlinear functions are not convex.

The approach used to identify the structure of the least-cost model was, first, to establish priorities for the structure of the model and, second, given these priorities, to investigate what was possible. The following priorities were established for the structure of the model: linear programming model, mixed integer linear programming model, and nonlinear programming model, in that order. Thus, the approach taken assumed a linear programming structure and proceeded to determine if that structure could accommodate the physical and biological processes associated with fish life-cycles and with the management of water resources in the Columbia River Basin. This required developing all the relationships in the model to determine if they were linear. For those relationships that were not linear, it required specifying the assumptions that were necessary to linearize the (nonlinear) relationships or to eliminate the nonlinear relationships altogether. In those cases where it was not possible to linearize a particular nonlinear relationship, consideration was given next to the use of integer variables to eliminate that nonlinear relationship. The use of nonlinear programming was considered only as a last resort.

The least-cost model developed in this chapter is described by “module.” A module for this purpose is defined as a set of activities performing similar functions. There are seven modules in all. The first three modules pertain to the hydrosystem. They are called “storage”, “hydropower”, and “irrigation”, respectively. The last four modules concern the fish life-cycle and fish migration. They are called "smolt production”, "smolt migration”, “ocean harvest and survival”, and “upstream migration of adults,” respectively. The last four modules depend to a large extent on the outputs of the first three. A conceptual diagram of the relationship between the fish life-cycle and the fish production and migration modules is provided in Figure 3.2.

The modules are organized functionally rather than geographically. As a result, individual dams and individual impoundments in the basin may be included in more than one module, and a particular module may include several dams and several impoundments. For example, the smolts described in the smolt migration module will traverse a number of different dams and impoundments along their way to the estuary; therefore, that module will include several dams and several impoundments.

The rationale for organizing the model functionally rather than geographically is that it is possible to construct generic modules that can be applied to more than one structure. For example, the form of the model used for irrigation water withdrawal is the same regardless of where the withdrawal occurs in the basin. While withdrawals at Grand Coulee may be vastly larger than those on small tributaries, they are modelled in much the same way, adjusting coefficients as needed to allow for features of particular withdrawal sites. Similarly, the structure of the model for hydroelectric production at the different run-of-the-river dams is the same regardless of the dam, and so hydropower also has its own module.

In addition to being divided functionally into modules, the least-cost model is also divided temporally into time periods. The overall time horizon for the model is fifteen years, for the following reasons. It is well-known that many runs, especially those of wild upstream stocks, are severely depleted in comparison to their historical levels forty to fifty years ago. Because of the many obstacles they face, including habitat limitations for spawning and rearing, downstream passage of smolts, ocean harvest, andupstream passage of potential spawners, it seems probable that it will take several generations for runs of these stocks to return to acceptable levels. Therefore, in principle at least, it is necessary to model the life cycle of stocks for at least two to three generations, of three to five years each (from spawning to returning adult spawners), in order to identify the

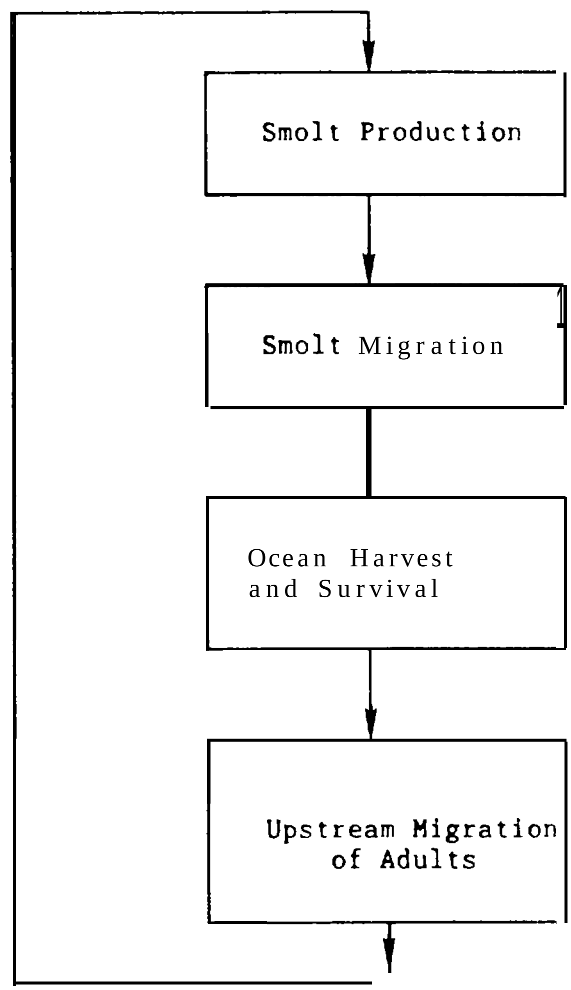

Figure 3.2. Relationship Between Fish Life Cycles and Fish Production and Migration Modules

management activities and the levels of those activities that will be needed over the next several years to return stocks to acceptable levels. As discussed in the next chapter, this poses a considerable challenge for existing mathematical programming algorithms and computers, although it is a problem that may be solvable by the use of advanced operations research techniques.

The fifteen year time horizon in the model is divided equally into fifteen years. Each year is further divided into twelve months. This level of disaggregation is needed in order to reflect temporal variability in the hydrologic cycle, in the demands for electricity and for irrigation water, in storage requirements at the storage reservoirs, and in salmonid life cycles and migration. While it might be desirable from a computational perspective to collapse the model into fewer years and fewer “months” per year, reducing the temporal resolution will invariably reduce the accuracy of the results. This is discussed in more detail in the next chapter.

Although the seven modules are described in more detail below, a brief synopsis of each module is presented in this introduction. The seven modules are divided into two groups. The first group is called the hydrosystem group. This group includes three modules. The second group is called the fish production and migration group. This group comprises four modules.

The hydrosystem group includes the storage, hydropower, and irrigation modules. The storage module manages storage and monthly releases at the major storage dams. This module is responsible for regulating flows in the system, subject to constraints on reservoir levels for flood control and recreation. The hydropower module produces electricity at the dams in the basin with significant generating capacity, using existing turbines and generators. This module is also subject to constraints, in this case on the hydraulic and electrical generating capacities of the turbines and generators at the various generating facilities. The irrigation module manages the withdrawal of irrigation water at various withdrawal sites in the basin. The irrigation module is subject to constraints on the physical capacities of installed equipment.

The fish production and migration group includes the smolt production, smolt migration, ocean harvest and survival, and upstream migration of adults modules. The smolt production module uses returning adult spawners and rearing habitat (either natural habitat or hatchery habitat) to produce smolts. Water is needed both at proper times and in proper quantities for this stage of the salmonids' freshwater life cycle, as it is for subsequent stages of the cycle. The water requirements for smolt production, for reasons that will be explained below, are handled indirectly as part of the habitat needs. The smolt migration module "transports" smolts downstream, from the locations where they are reared to the estuary below Bonneville Dam.

After the smolts arrive at the estuary below Bonneville Dam, they are "transferred" to the ocean harvest and survival module. This module, developed in another part of the project (see part III) describes the growth, natural mortality, and ocean harvest of salmon while they are in the ocean. The upstream migration of adults module accepts the survivors that escape the ocean and transports them to their respective spawning grounds where, in the smolt production module, they lay and fertilize eggs. In this module, adult fish are subjected to in-river harvest and both natural and dam induced mortality. The adults of most species die after spawning.

OVERVIEW OF MODEL

An example is presented in this section to illustrate how the modular structure is used to describe the life cycle of an actual stock. For this example, a didactic region containing two storage dams, two run-of-the-river dams, two irrigation withdrawal sites, and two salmonid stocks has been constructed. This region is shown in the map in Figure 3.3. The stocks are assumed to mingle in the ocean, so that the ocean fishery is a mixed-stock fishery, and their downstream and upstream migration periods are assumed to overlap. A further assumption is that all four dams in the didactic region

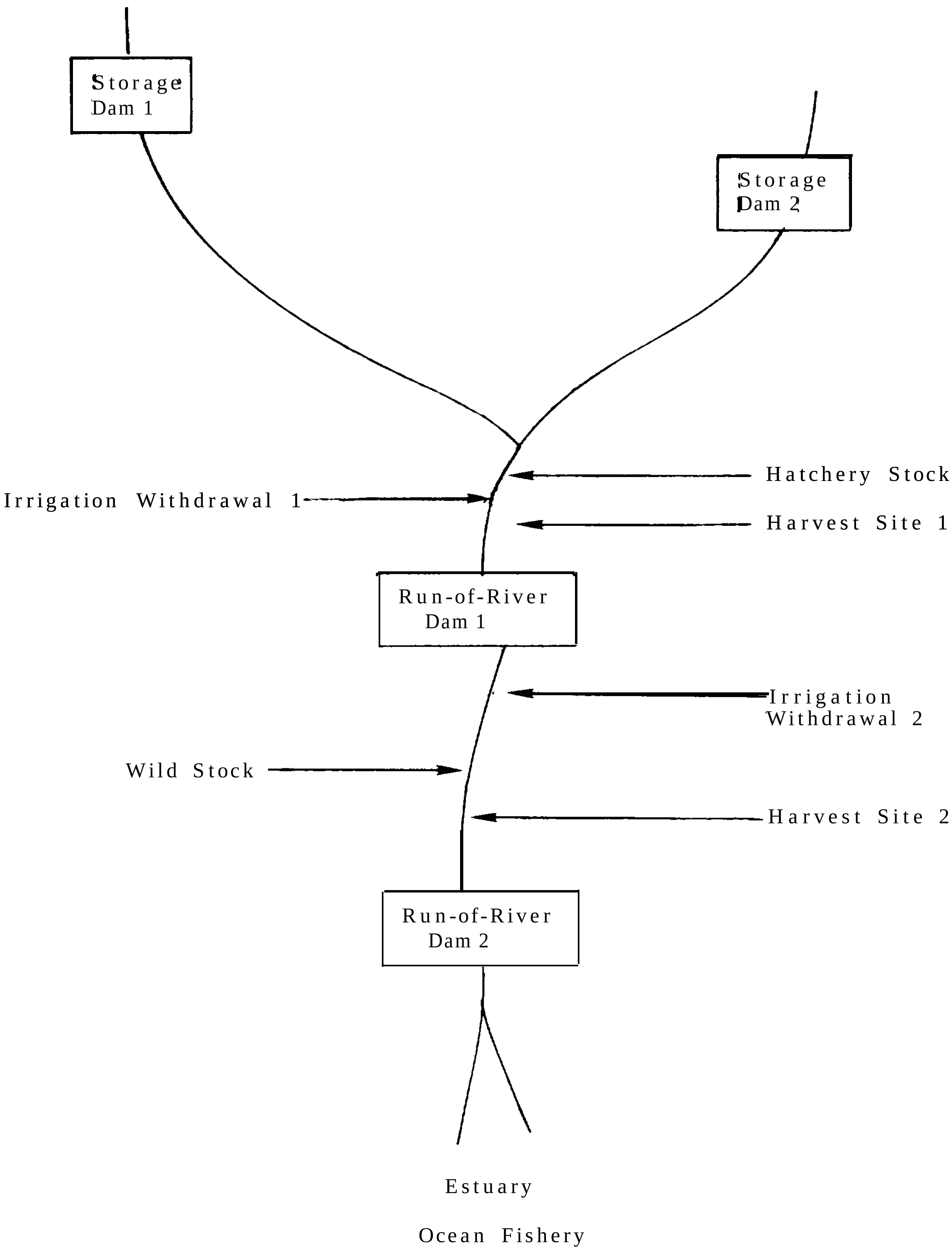

Figure 3.3. Map of Didactic Region

have significant hydroelectric generating capacity, and that all the dams are part of the same electricity distribution grid. Management activities are included in the modules for each month of the fifteen year planning horizon, with the exception of the irrigation module, as this module "operates" only during the summer irrigation season.

The storage module contains the management activities at the two storage dams in the didactic region that pertain to storage and releases, and to storage and release regulations (rule curves). These activities include tracking the inflows to each reservoir from upstream dams and unregulated tributaries, the reservoir levels at each dam to ensure that storage reservoirs do not overflow and that there is sufficient storage capacity available for flood control in appropriate months, and the quantities of water released each month.

The hydropower module includes all activities related to hydropower production at the four dams in the didactic region, including the quantities of water that pass through the turbines and that flow over the spillways, the levels of the reservoirs, and the amounts of electricity generated, both at particular dams and in total. As shown in more detail below, power production is modelled differently at the storage dams than it is at the run-of-the-river dams because at the storage dams the head varies from month to month whereas at the run-of-the-river dams the head is assumed to be constant.

The irrigation module includes the withdrawal of irrigation water at each of the two irrigation withdrawal sites in the didactic region.

The smolt production module includes spawning and rearing for both the wild (natural) stocks and the hatchery stocks in the didactic region. The inputs to this module include the number of adult spawners returning via the upstream migration of adults module (described below) and the amount of spawning habitat. The outputs from this module are the number of smolts of each stock. The activities in this module are described by stock to permit analyzing the impacts on individual stocks.

The smolt migration module involves a somewhat broader range of inputs. They include smol ts “produced” in the smolt production module; stream flows (net of irrigation water withdrawals) along the migration routes provided by the storage module; and the proportion of total flow passing through the turbines and over the spillway, at each dam encountered by each stock of migrating smolts, from the hydropower module. The smolt migration module includes smolt collection and diversion structures and other transport options such as trucking. The principal output of the smolt production module is the number of smolts (of each stock) arriving at the estuary below the lower-most dam in the didactic region, along with the timing of their arrival.

The ocean harvest module receives smolts from the smolt migration module and accounts for fish losses in the ocean due to natural mortality and to ocean harvest up to the point where adult fish are ready to return to the river to spawn. The ocean harvest module also accounts for economic gains to the ocean fishing industry from ocean harvest.

The upstream migration of adults module receives survivors that escape the ocean and return to the river to spawn, and routes them up the river past the two run-of-the-river dams to their natural spawning grounds or hatcheries, subject to harvest at each of two locations and to natural and dam induced mortality. The returning adult fish become available to the smolt production module as adult spawners. The map in Figure 3.3 also shows how the various modules relate to geographic features of the didactic region.

As mentioned above, the model is divided temporally as well as functionally. The model extends over a fifteen year period, tracking hydrosystem operations and the life cycle of salmonid on a monthly basis for each year of the fifteen year period. While a monthly level of temporal resolution in the life cycle of fish is not needed for the two stocks in this didactic example, it will be needed for the full scale model (to be developed in the next phase of the project) when 30 or more stocks are included, since this number represents a wide variety of salmonid stocks and life cycles where significant activities can be expected to occur in every month of the year. Adults of different stocks are in the mainstem river almost year-round, and although smolts migrate primarily in the spring, juveniles of different stocks are in the mainstem river in all months; hence, they are subject to variations in stream flow year-round. While it would be desirable from a computational perspective to keep the model as small as possible, there are trade-offs between model size and the probability of identifying least-cost management strategies that need to be considered.

The remainder of this chapter is devoted to describing the seven modules discussed briefly above. Chapter 4 addresses the computational problems that may arise as a result of the large size. It also discusses approaches to coping with large model size, including reducing the size of the model (by aggregating activities), decomposing the model into parts that can be solved sequentially rather than simultaneously, and using super computers for large models.

The least-cost model described in this chapter is based on theoretical and empirical relationships derived in part from the results of on-going or recently completed research in the Pacific Northwest and in part from the fisheries simulation model developed in another part of this project (Lee, 1987). Although the relationships used in developing the least-cost model seem reasonable based on the information that was available, it is not known for certain that they are the best representations available of the true relationships.

DESCRIPTION OF THE HYDROSYSTEM MODULES

The least-cost model is divided into seven modules, as described above. The first three modules--storage, hydropower, and irrigation--pertain to the water resources of the Columbia River system. These three modules are described in this section. The last four modules pertain to fish life cycles. They are described in the next major section.

The description of each of the hydrosystem modules is organized around seven sections:

- Introduction (to each module).
- Inputs required from other modules.
- Outputs prepared for other modules.
- Constraints on levels of the management and state variables in the module.
- Mathematical relationships in the module.
- Costs of fish mitigation alternatives, including the opportunity costs of nonstructural alternatives such as increased stream flows and modification of harvest management practices for adult fish.
- Assumptions required to structure the model as a linear programming (LP) model or as a mixed-integer LP model.

The descriptions of modules in this section concentrate mainly on the relationships among activities that occur within the same time period (i.e., month). Interactions across time periods are not given much attention. For example, the method used to track capital installed in one time period across subsequent time periods is not discussed. Rather, the discussions focus on the methods used to incorporate what is known about hydrosystem operations and salmonid life cycles within a single time period. The rationale for focusing on a single time period is that the structure of the least-cost model is determined by the relationships among the management and state variables in the same time period. The number of time periods in the model affect its size, but not its structure.

In the discussions of the modules below, both intra (within) module and inter (among) module transfers and interactions are indicated. Therefore, it is necessary at this point to define some of the management and state variables used in the least-cost model. The inter--module variables (the inputs to and the outputs from the different modules) are listed in Table 3.1. The intra-module variables are defined in the text as they are used in the mathematical development. The three hydrosystem modules are described in the next three sections.

Storage Module

The first hydrosystem module is the storage module. This module describes the operations of the principal storage reservoirs in the U.S. portion of the Columbia River Basin. (The storage reservoirs in the Canadian portion of the basin are not incorporated in the least-cost model). These storage reservoirs serve a number of beneficial purposes, including the provision of flat-water recreation, storage capacity for flood control, and both storage and (hydraulic) head for the production of hydropower.

The impacts of the storage reservoirs on the salmonid populations in the basin have been quite severe. The impoundments have flooded many spawning areas, slowed both downstream and upstream passage of migrating fish, and in some cases blocked upstream passage of salmonids altogether. The role of storage reservoirs in flow regulation is given considerable attention in the development of the model since it is primarily in this area that possibilities for mitigating their impact on fish populations exist.

Storage dams and their impoundments are modelled as variable-head reservoirs whose principal purpose is to generate hydropower and to regulate flows to the free-flowing reaches and run-of-the-river reservoirs downstream from them, subject to constraints on their operations for flood control and for flat-water recreation.

Inputs. There are two principal inputs to the storage module: monthly inflows from unregulated tributaries and monthly inflows from storage reservoirs upstream of the reservoir in question. In addition, since storage reservoirs have a large storage capacity relative to mean monthly inflows and

Table 3.1. Definitions of Inter-Module Variables

Variable	Definition of Variable
Hydrosystem Variables	
$E_{i,t}$	Electricity produced at dam i, in time period t, in kilowatt-hours (KWH)
$QS_{i,t}$	Flow over spillway at dam i, time period t, in cubic feet per second (CFS)
$QG_{i,t}$	Flow through turbines (used for electrical generation) at dam i, time period t, in CFS
$QT_{i,t}$	Total flow past dam i, time period t, in CFS ($QT_{i,t} = QS_{i,t} + QG_{i,t}$)
$OF_{i,t}$	Total outflow from storage reservoir i, time period t, in acre-feet (AF)
$RIF_{i,t}$	Regulated inflow to run-of-the-river reservoir i, time period t, in AF
$UIF_{i,t}$	Unregulated inflow to run-of-the-river reservoir i, time period t, in AF
$TF_{i,t}$	Total river flow, location i, time period t, in AF ($TF_{i,t} = RIF_{i,t} + UIF_{i,t}$)
$IRW_{i,t}$	Irrigation water withdrawal at site i, time period t, in AF
$IRR_{i,t}$	Irrigation water return flow at site i, time period t, in AF.
Fish Migration Variables	
$ADS_{i,t,s}$	Adult spawners, location i, time period t, species s, in number of fish.
$ADR_{i,t,s}$	Adults returning to estuary, location i, time period t, species s, in number of fish.
$SMOLT_{i,t,s}$	Smolts, location i, time period t, species s, in number of smolts.

outflows, the level of storage at the end of the previous month is also an input.

<u>outputs</u>. The output from the storage module is a regulated outflow. This outflow is computed as the monthly inflow to the reservoir net of changes in storage and of evaporation and rainfall in that month. The change in storage in a particular month can be positive or negative.

<u>Constraints</u>. The storage reservoirs operate under a set of constraints. These constraints are of two types--physical and operational. The physical constraints are based on the physical characteristics of the dams and on the topographic configuration of their impoundments. They include the maximum and minimum capacities of the reservoirs, and the hydraulic capacities of the turbines, diversion channels, and spillways. The physical constraints are assumed to be the same for all time periods in the model. The operational constraints are usually referred to as “rule curves”.

The rule curves for a particular reservoir reflect tradeoffs and compromises between the various competing uses of the impoundment, as noted above. For example, the risk of flooding could be minimized if the reservoirs were empty much of the year except when the natural unregulated flow would result in flooding downstream. During these periods of high flow, the reservoirs would be used to store the excess inflows, permitting outflows only to the extent that they would not cause flooding downstream. Following the flood season (which occurs in the late spring), the stored flood flows would be released gradually over the next several months so that the reservoirs would be empty (at the level of dead storage) prior to the flood season the following spring.

In contrast to flood control, the best management practice for flat-water recreation would be to maintain reservoirs at approximately the same level year-round. This would eliminate excessively high water levels, mud fiats, and other impediments to recreation caused by fluctuations in reservoir levels. For hydropower production it might be best to maintain reservoir levels as high as possible at all times since hydropower generation is proportional to the product of the flow through the turbines and the hydraulic head (the difference in elevation between the surface of the reservoir and the surface of the tail-water, or the center line of the turbine). Note that this would be strictly true only for turbines designed to operate most efficiently at maximum head, and there were a market for all the power generated.

In the first phase of model development, the existing reservoir rule curves will be taken as given. In later phases of model development, the rule curves will be relaxed to permit exploring trade-offs between improvements in fish runs and reductions in some of the benefits the rule curves were originally designed to provide, such as an assured supply of flood control capacity. From a computational perspective, this means changing the minimum and maximum target flows and target storage levels specified by the existing rule curves at the expense of increasing the risk and magnitude of floods that the rule curves were designed to mitigate. Thus, the operational constraints in the model will be relaxed in order to allow it greater latitude in identifying least-cost mitigation strategies for increasing fish runs.

<u>Mathematical Relationships</u>. The principal function of the storage module is to regulate the monthly outflows and storage levels at the major storage dams in the Columbia River Basin. The following equation expresses the mass balance of water entering and leaving the reservoir in a particular month.

$$S_{i,t} = S_{i,t-1} + RIF_{i,t} + UIF_{i,t} - EVAP_{i,t} - OF_{i,t} \qquad (3\text{-}1)$$

where $S_{i,t}$ = storage in reservoir i at the end of time period t, in acre-feet (AF);

$RIF_{i,t}$ = total regulated inflow to reservoir i from upstream storage reservoirs in time period t, in AF;

$UIF_{i,t}$ = total unregulated inflow to reservoir i from unregulated tributaries in time period t, in AF;

$EVAP_{i,t}$ = total evaporation at reservoir i during time period t, in AF;

$OF_{i,t}$ = total outflow from reservoir i in time period t, in AF.

There are several constraints on the operation of storage reservoirs, some reflecting physical limitations, others reflecting operating characteristics (rule curves). The first set of constraints pertains to storage requirements and capacities.

$$S_{i,t} \leq SCAP_i \tag{3-2}$$

where $SCAP_i$ is the maximum capacity of reservoir i. For the first phase of model development where the rule curves will be taken as given.

$$S_{i,t} \geq STARMIN_{i,t}$$

$$S_{i,t} \leq STARMAX_{i,t} \tag{3-3}$$

where $STARMIN_{i,t}$ = minimum target storage level in reservoir i at the end of time period t, in AF;

$STARMAX_{i,t}$ = maximum target storage level in reservoir i at the end of time period t, in AF.

Finally, for both the first and subsequent phases of model development,

$$OF_{i,t} \leq OFMAX_i \tag{3-4}$$

where $OFMAX_i$ = maximum permissible combined total outflow through the turbines and over the spillway for flood control.

For a set of activities that controls much of what happens in the least-cost model, the constraint set for the storage reservoirs is quite compact. As will be seen from the descriptions of the other modules, however, the regulated outflows from the storage reservoirs are the principal inputs to some of the other modules, and it is in these modules that some of the complications arise.

Costs. There are no costs in the storage module. No mitigation structures are "built" in this module and there are no increases in the costs of operating the reservoirs. However, there are "opportunity costs" associated with relaxing the rule curves that are not reflected in the model or analysis. These include losses caused by floods and losses to recreation due to fluctuations in reservoir levels.

Assumptions Required for Linear Program. No assumption is required to permit incorporating this module within an LP framework. As shown by Eqs. (3-l) through (3-4) the mass balance relationship and all the constraining relationships are linear. The only assumption of note concerns the choice of a monthly time step for the inflows and outflows, for changes in storage, and for the constraints on reservoir levels. This time step does not capture variations in the system of shorter duration than one month.

Hydropower Module

The hydropower module includes the major dams in the U.S. portion of the Columbia River Basin that have significant hydroelectric generating capacity. There are two types of hydroelectric dams. One type is the run-of-the-river dams. This type is characterized by a fixed head and insignificant storage capacity. The other type is the storage dams. This type is characterized by a variable head and significant storage capacity. These two types of dams are modelled differently because of differences in the treatment of head. Head is an important factor in determining the amount of electricity generated for a given flow through the turbines and in one type of dam the head varies from month to month and in the other type it does not.

Inputs. Each dam in the hydropower module requires inputs of monthly flows. In the case of the variable-head storage reservoirs, these flows are "provided" by the storage module. In the case of the run-of-the-river dams, the inflow is the sum of the flows from the dams immediately upstream plus the flows from all the intervening unregulated tributaries, less evaporation. There may be more than one upstream dam if the run-of-the-river dam in question is the first dam below the confluence of two major regulated tributaries.

outputs. The outputs from the hydropower module are the amounts of electricity generated, both for each dam and in total for the region, and the flows at each dam through the turbines, over the spillways, and in total. The flows over the spillways, while not important to electricity production *per se*, are important for the downriver migration of smolts.

Constraints. The hydropower module operates under several sets of constraints. The first set concerns the maximum electricity generating capacity at each dam. This is the maximum capacity of the generators. It cannot be exceeded due to physical limitations. The second set of constraints concerns the maximum hydraulic capacity of the turbines (the maximum flow possible through the turbines). The last set of constraints involves the minimum electricity requirements, or firm power. In the analysis, either there will be a constraint on the minimum amount of electricity produced in the region each month or there will be constraints on the minimum amount of electricity produced at each dam.

Mathematical Relationships. The principal activity in this module concerns the amount of electricity produced at each dam each month. The constraints involve minimum and maximum limits on the generation of electricity, both at particular dams and in total for the region. There are also mass-balance relationships to ensure that the inputs of water equal the outputs.

The first relationship concerns the generation of electricity. This may be expressed as the product of the flow through the turbines and the head.

$$E_{i,t} = K_{i,t} * QG_{i,t} * H_{i,t} \tag{3-5}$$

where $E_{i,t}$ = total amount of electricity generated at dam i, time period t, in KWH

$QG_{i,t}$ = flow through the turbines at dam i, time period t, in CFS

$H_{i,t}$ = hydraulic head at dam i, time period t, in feet

$K_{i,t}$ = conversion factor, reservoir i, time period t, reflecting the efficiencies of both the turbines and generators.

For the run-of-the-river dams, $H_{i,t}$ is assumed to be a constant. Expressed mathematically,

$$H_{i,t} = H_i^*$$

where H_i^* represents the fixed head at dam i.

For the storage dams, $H_{i,t}$ will vary, but it will be constrained by the rule curves.

$$H_{i,t} \leq HMAX_i \tag{3-6}$$

and $$H_{i,t} \geq HMIN_i \tag{3-7}$$

where $HMAX_i$ = maximum hydraulic head at dam i due to rule curve constraints, if any, or to the maximum level of the reservoir.

and $HMIN_i$ = minimum hydraulic head at dam i due to rule curve constraints, if any.

Note that Eq. (3-5) involves the product of two decision variables, flow (QG) and head (H), for storage dams. For LP and mixed integer LP models, multiplicative terms are not permitted. In order to avoid multiplicative terms, it is necessary to formulate "discrete" integer values for H_t. One way of doing this is to divide $H_{i,t}$ and $S_{i,t}$ into N intervals where

$$S_{i,t} = \sum_{n=1}^{N} S_{i,t,n} \tag{3-8}$$

$$H_{i,t} = \sum_{n=1}^{N} H_{i,t,n} \tag{3-9}$$

and where $S_{i,t,n}$ = storage interval n at reservoir i in time period t, in AF. (The division of the active storage into discrete intervals is needed in order to divide the head into corresponding discrete intervals.)

$H_{i,t,n}$ = hydraulic head interval n at reservoir i in time period t, in feet.

$S_{i,t}$ = total active storage at reservoir i in time period t, in AF.

$H_{i,t}$ = total hydraulic head at reservoir i in time period t, in feet.

Next, relate $H_{i,t,n}$ to $S_{i,t,n}$ based on the topography of the land under the impoundment and the volume of storage.

$$H_{i,t,n} = b_{i,n} * S_{i,t,n} \tag{3-10}$$

where $b_{i,n}$ relates storage interval n to head interval n. Note that $H_{i,t,n}$ is >0 if and only if $S_{i,t,n} > 0$. Since from Eq. (3-9) the total head is equal to the sum of the intervals, the generation of electricity may be expressed as

$$E_{i,t} = K_{i,t} * QG_{i,t} * H_{i,t,n} \tag{3-11}$$

This will require N different generating activities for each storage reservoir where the head "chosen" for the generation of hydropower depends upon the "interval" of storage in that month. This requires the use of integer variables and integer programming. The portion of the mixed integer LP tableau used for the generation of hydropower at the variable head storage reservoirs is shown in Table 3.2.

There are several constraints in the hydropower module. In the first phase of model development, the total amount of electricity generated in the system is assumed to be constrained. This may be expressed as

$$\sum_{i=1}^{I} E_{i,t} \geq ET_t \tag{3-12}$$

where ET_t = total electricity required in the grid (or region) in time period t, in KWH. Note that this constraint allows the model to choose the dams where the electricity will be generated. In subsequent phases of model development, given prices for electricity, the constraint on the total electricity generated will be reduced to cover only the base load and the electricity produced over and above the base load will be "priced" in the objective function. These equations are not shown.

The remaining constraints in the hydropower module involve the maximum flow through the turbines in time period t. The first constraint ensures that the flow through the turbines, QG, does not exceed the total flow past the dam, QT.

$$QG_{i,t} \leq QT_{i,t} \tag{3-13}$$

The second constraint ensures that the flow through the turbines does not exceed the maximum hydraulic capacity.

$$QG_{i,t} \leq QGMAX_i \tag{3-14}$$

Table 3.2. The Mixed Integer Linear Programming Tableau Used for the Generation of Hydropower at the Variable Head Storage Reservoirs

Rows	Columns									
Row Name	Column									RHS
	Q_{tot}	Q_{stor}	Q_{tu}	Q_{sp}	ST_{t-1}	HCOL1	HCOL2	GEN1	GEN2	
FLOWROW	+1	-1	-1	-1	0	0	0	0	0	=0
STORROW	0	+s1	0	0	+1	$-C_1$	$-C_2$	0	0	≥ 0
TURBROW	0	0	+1	0	0	0	0	-1	-1	≥ 0
HROW1	0	0	0	0	0	$+10^6$	0	-1	0	≥ 0
HROW2	0	0	0	0	0	0	$+10^6$	0	-1	≥ 0
ELECROW	0	0	0	0	0	0	0	$+K_1$	+K2	≥ 0
HTOT	0	0	0	0	0	+1	+1	0	0	≤ 1

Definitions:

Columns:

Q_{tot} = Total inflow, cfs

Q_{stor} = Change in storage, AF

Q_{tu} = Outflow through turbines, cfs

Q_{sp} = Outflow over spillway, cfs

ST_{t-1} = Storage level from previous month, AF

HCOL1 = Integer (0,1) variable, 1 if and only if STORROW can be > 0

HCOL2 = Integer (0,1) variable, 1 if and only if STORROW can be > 0

GEN1 = Generating activity 1

GEN2 = Generating activity 2

Rows:

FLOWROW = Inflow summing row, constrained to be equal to 0

STORROW = Storage summing row, constrained to be greater than 0

TURBROW = Turbine flow summing row, constrained to be equal to 0

HROW1, HROW2 = Summing rows for head/generation column linkages

ELECROW = Electricity production summing row

Table 3.2 Continued

HTOT = Constraint row to ensure that HCOL1 and HCOL2 do not operate simultaneously

Non-Zero coefficients not equal to $\pm$ 1:

S_1 = Relates flows (in cfs) to storage (in AF), in AF/cfs

C_1, C_2 = Relates head (in feet) to storage (in AF), in feet/AF

K_1, K_2 = relates power generation (in MW) to flows, in cfs, in MW/cfs

10^6 = These "large" coefficients are chosen to provide "capacity" for the generation activities, GEN1 and GEN2.

Note that $C_1 < C_2$ and $K_1 < K_2$. The result is that although HCOL2 requires more storage (and hence a higher head) to operate than does HCOL1, GEN2 will produce more electricity per unit of flow than will GEN1, since K_2 is greater than K_1.

where $QGMAX_i$ represents the maximum hydraulic capacity of the turbines at dam i, in CFS.

Costs. There are no costs in this module other than the "opportunity costs" associated with the electricity production foregone, but these costs are potentially very high. The costs of passage enhancement for downstream smolt migration and upstream adults migration are included in the respective fish migration modules discussed below.

Assumptions Required for Linear Program. The hydropower module requires the use of integer variables and a number of simplifying assumptions to permit incorporating it within an LP framework. The integer variables are required at the variable head storage dams to permit treating the hydraulic head as a constant rather than as a variable, and thus to eliminate the product of two variables in the cost (objective) function. The assumptions made are summarized below.

- Constant (fixed) head at the run-of-the-river dams.
- Variable head at the storage dams.
- Efficiencies of turbines and generators at both the fixed and variable-head dams do not vary with flow or head. (The coefficient, $K_{i,t}$, in Eq. (3-5) can be specified by time periods to allow for changes in efficiency.)
- No change in the design or capacities of dams, turbines, or generators during the study period.
- Sufficiently accurate results can be obtained using integer variables for the head-flow relationships described above.

Irrigation Module

Irrigation water withdrawals also cause problems for salmonids due to their effects on the quantities and timing of stream flows. The impacts on salmonids are included in the fish production and migration modules described below.

The irrigation module is concerned principally with the effects on stream flows of water withdrawals for irrigation. In principle the irrigation module would include one set of management activities for each of the major irrigation diversions, for each time period in the model. However, since irrigation is a seasonal activity, it is not necessary to incorporate irrigation activities in the module for months in which little or no irrigation actually takes place.

Inputs. There are two principal inputs to the irrigation module. The first input is the stream flow at the irrigation water withdrawal sites. The second input is the electricity used to pump irrigation water. While electricity is not needed for the pump stations powered by diesel or gasoline engines, present understanding is that the major irrigation sites use electricity and that this usage is a significant portion of the total hydropower generating capacity in the basin, on the order of 10 percent of the total electricity generated annually.

outputs. The principal outputs of the irrigation module are the quantities and timing of irrigation water withdrawals, the quantities and timing of electricity consumed in those withdrawals, and the impacts on stream flows.

Mathematical Relationships. The principal activity in the module is the withdrawal of irrigation water to meet irrigation needs. The irrigation water "requirements" are constrained and the "demands" are priced.

The energy and electricity requirements for irrigation water withdrawals are expressed next. The energy required to withdraw the water from the river may be expressed as

$$ERS_{i,t} = K1 * H1_{i,t} * IRW_{i,t} \qquad (3\text{-}15)$$

The amount of electricity required for this may be expressed as

$$IRELEC_{i,t} = b_i * ERS_{i,t} \tag{3-16}$$

where

$ERS_{i,t}$ = theoretical energy required at withdrawal site i in time period t to pump irrigation water at a fixed hydraulic head (H1) for subsequent distribution, in foot-pounds.

$IRW_{i,t}$ = quantity of irrigation water withdrawn from the river at location i in time period t, in AF.

$H1_{i,t}$ = hydraulic head at withdrawal site i in time period t, in feet. (This is a parameter in the analysis, not a variable.)

K1 = factor used to convert acre-feet of water to pounds of water.

$IRELEC_{i,t}$ = electricity required to pump irrigation water (includes the efficiencies of pumps and motors), in KWH.

b_i = coefficient used to convert theoretical energy in foot-pounds to the amount of electricity consumed, in KWH. This coefficient includes the efficiencies of the pumps and motors. Note that this coefficient may be zero (for diesel or gasoline powered pumps).

There are several constraints in the irrigation module. First, the total electricity used for pumping irrigation water must be less than the total produced in the region.

$$\sum_{i=1}^{M} IRELEC_{i,t} \leq \sum_{i=1}^{N} E_{i,t} \tag{3-17}$$

where M is the number of irrigation water withdrawal sites in the model and N is the number of hydroelectric dams in the model.

The irrigation water withdrawal at each site cannot exceed the capacity of the irrigation pumps.

$$IRW_{i,t} \leq IRMAX_i \tag{3-18}$$

where $IRMAX_i$ = capacity of pumps at withdrawal site i, in AF

The irrigation water withdrawal at each site cannot exceed total flow in the river at that location.

$$IRW_{i,t} \leq TF_{i,t} \tag{3-19}$$

For the first phase of model development, the irrigation water withdrawals are constrained to be met. This may be expressed as

$$IRW_{i,t} = IRDEM_{i,t} \tag{3-20}$$

where $IRDEM_{i,t}$ represents the irrigation requirement at withdrawal site i in time period t, in AF.

In subsequent phases of model development, the irrigation water withdrawal will be constrained, but it will also be priced. Using the notation above, this may be expressed as

$$IRDEM_{i,t} = IROLD_{i,t} - IRRIGHT_{i,t} \tag{3-21}$$

$$IRCOST_{i,t} = IRRIGHT_{i,t} * C_{i,t} \tag{3-22}$$

where,

$IROLD_{i,t}$ = original irrigation water withdrawal, in AF.
$IRRGHT_{i,t}$ = water rights purchased from irrigation, in AF.
$IRCOST_{i,t}$ = total cost of irrigation water rights purchased, in dollars.

$C_{i,t}$ = price of irrigation water rights at withdrawal site i in time period t, in dollars per AF.

Costs. The only cost in this module is the "opportunity cost" associated with withdrawing water from irrigation to increase instream flows for fish.

Assumptions Required for Linear Program. No assumption is required to permit incorporating this module within an LP framework. As shown by Eqs. (3-17) through (3-22), the mass balance relationships and constraints are all linear. The main assumption in this module is that return flows from irrigation are not significant. If this proves to be incorrect, it is relatively easy to include them in an LP structure. The only potential problem is that the return flows may occur far downstream from the withdrawals and perhaps also in a later time period. Although this may affect model accuracy (if the timing and amounts of return flows are not known precisely), it has no effect on model structure.

This concludes the description of the three hydrosystem modules. The next section describes the four fish production and migration modules.

DESCRIPTION OF THE FISH PRODUCTION AND MIGRATION MODULES

The four fish production and migration modules are described in this section. These include the smolt production module, the smolt migration module, the ocean harvest and survival module, and the upstream migration of adults module. The relationship between these four modules and the fish life cycle is shown schematically in Figure 3.2.

Smolt Production Module

The smolt production module includes a number of functions. Specifically, it includes the spawning of returning adults, the hatching of eggs, the production of fry, and the production of smolts, at both the hatcheries and the natural spawning and rearing areas in the basin. The same model structure can be used for both, although the values of the coefficients in the model will vary greatly. Variations in coefficients in the model account for such factors as the vastly higher productivity of spawners at hatcheries and the fact that hatcheries incur capital and operating costs not associated with existing natural spawning and rearing areas. The role of instream flows is also different for the two types of smolt production since natural spawning and rearing areas depend directly on instream flows and hatcheries depend only indirectly on instream flows. Hatcheries provide a more controlled environment in which spawning and rearing takes place. As with the hydrosystem modules described above, the smolt production module has a set of activities for each spawning and rearing area, for each time period in the model. In addition, the smolt production module is further disaggregated into species which may spawn simultaneously at the same location. For purposes of this model, the identification of fish by species, hatchery or natural production, spawning location, and spawning time (month) is sufficient to identify a particular stock. That is the level of resolution here. Note that the relationships between spawning, flow, and habitat are not well understood, and the equations outlined here should be viewed as tentative and subject to change as more information becomes available.

The functional relationships in the smolt production module are more complex than those in the hydrosystem module. For this reason they are discussed in detail in this introduction. This discussion is not intended to be a comprehensive survey of the many models and studies of smolt production in the Pacific Northwest. Rather it is an overview of material needed to understand the particular functional relationships chosen to describe smolt production in the least-cost model.

The smolt production module “produces” two outputs for each stock in the model. These outputs are the number of eggs and the number of smolts. For purposes of this model, eggs are defined to be viable fertilized eggs; smolts are defined to be immature salmonids ready to begin their downstream migration to the ocean. This definition of "smolt" is not universally accepted among fisheries biologists, but it will serve for purposes of the least-cost model. Smolts produced in this module are passed to the smolt migration module described later.

Egg Production. The production of eggs may be expressed as a function of two variables, the number of adult spawners and the spawning habitat.

$$EGGS = f\ (ADS,\ HAB) \tag{3-23}$$

where,

EGGS = Number of eggs produced by a given stock.

ADS = Number of returning adult spawners.

HAB = Available spawning habitat

The spawning habitat is described in more detail below.

Two simplifying assumptions are made for purposes of the model. First, the spawning habitat is not a limiting factor for the stocks in the region. (Currently, there are not enough adult spawners to use up all the available spawning areas in most subbasins.) Second, for a given stock, the average number of eggs produced per adult spawner is a constant so that the total number of eggs produced may be expressed as a linear function of the number of adult spawners. By assumption the spawning habitat is not limiting. Therefore, Eq. (3-23) may be simplified to

$$EGGS = f\ (ADS) \tag{3-24}$$

The assumption of a constant number of eggs produced per adult spawner leads to the following linear expression for the production of eggs.

$$EGGS = c * ADS \tag{3-25}$$

where EGGS and ADS are defined above and "c" is a stock-specific constant expressed in number of eggs per adult spawner.

Smolt Production. The equation for the production of smolts can be developed in a similar fashion. The number of smolts produced is a function of the number of viable eggs, Eq. (3-25), and the availability of suitable rearing habitat. (The definition of rearing habitat is provided below.). Unlike spawning habitat which is assumed not to be constraining, rearing habitat is assumed to be a potential limiting factor in the production of smolts. Therefore, it is included in the equation for smolt production.

The availability of rearing habitat is assumed to be a function of two variables, stream flow and the quantity of “substrate”. Stream flow is self explanatory. Substrate includes such things as stream bed composition, vegetation, cover, and food supply. The concept of substrate is deliberately left quite general since improvements in factors affecting available rearing habitat (other than stream flow) are specific to particular spawning sites and stocks. Because the availability of rearing habitat can be a limiting factor in smolt production, the model includes activities to expand it both by increasing regulated flows (where rearing occurs at sites affected by dams in the hydrosystem module) and by additions to the “substrate”.

Before continuing with the development of a mathematical expression for the production of smolts, a complication needs to be mentioned. The production of smolts from eggs is assumed to follow what economists refer to as declining economies of scale with respect to eggs, assuming the availability of habitat is fixed. Fisheries biologists usually call this a Beverton-Holt curve. The shape of this function is shown in Figure 3.4. While this shape is relatively easy to accommodate in a linear program, an explanation of its implications is necessary.

The most important implication of the shape of the relationship shown in Figure 3.4, in contrast to egg production, is it is nonlinear given a constant amount of rearing habitat. The reason for this is the following. As the rearing habitat becomes more crowded, its ability to support the "last" egg from hatching through to smoltification is reduced due to competition for scarce resources, such as food and cover. This means that there is an absolute upper bound, approached asymptotically, to the number of

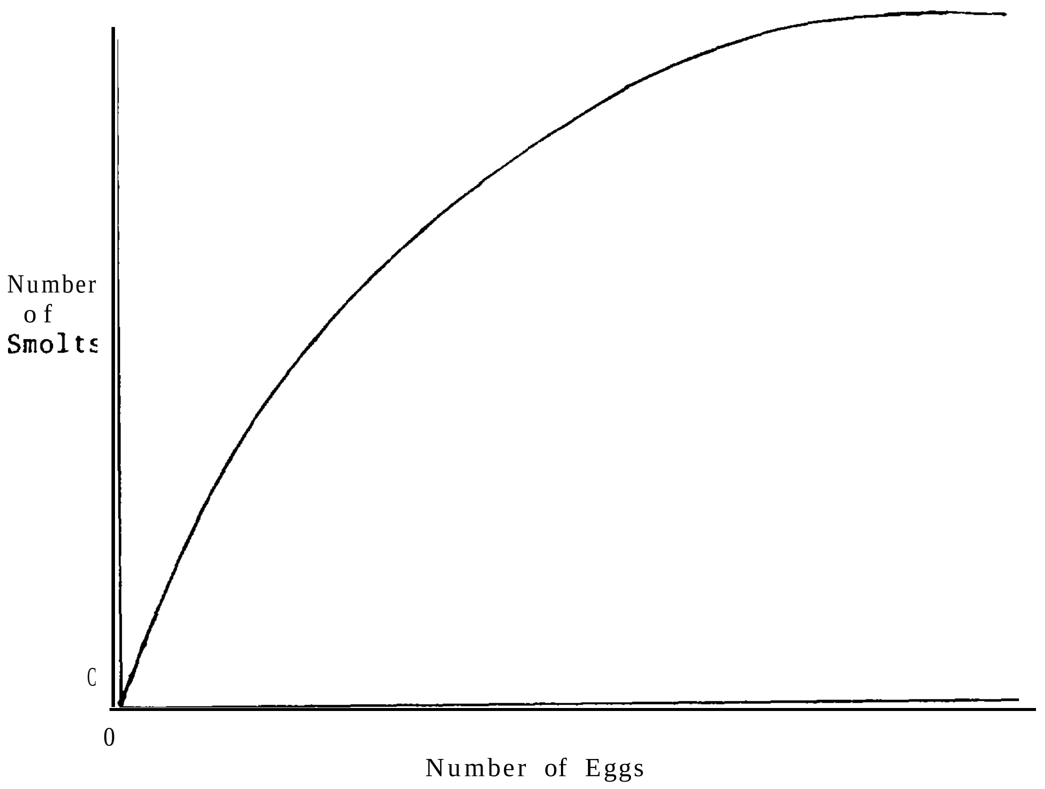

Figure 3.4. Relationship Between the Number of Eggs and the Number of Smolts Produced

smolts that can be produced in a given amount of rearing habitat regardless of the number of eggs available.

This relationship can be expressed mathematically as:

$$SMOLTS = f\,(EGGS, HAB) \tag{3-26}$$

where SMOLTS = number of smolts produced.

EGGS = number of eggs available (from Eq. (3-25)).

HAB = quantity of rearing habitat available.

For purposes of this development, a unit of rearing habitat is defined as the quantity of habitat that can support the rearing of 1,000 smolts at maximum carrying capacity.

The next step is to incorporate the declining marginal productivity of eggs in the smolt production relationship, Eq. (3-26). In this development, it is assumed for simplicity that a fixed amount of habitat is available, say one unit. In the following development, this assumption is relaxed.

First, divide the total number of eggs available into intervals, say three, corresponding to three levels of declining marginal productivity. This is shown in Figure 3.3. The total number of eggs available may be expressed as

$$EGGS = EGG_1 + EGG_2 + EGG_3 \tag{3-27}$$

where EGG_i, $i=1,2,3$, represent the number of eggs in the three productivity intervals.

The three subgroups of eggs shown in Eq. (3-27) are composed of identical eggs. They are divided in this fashion only to account for declining marginal productivity due to crowding. The next step is to express the equation for the total number of smolts produced by these three groups of eggs.

$$SMOLT = a1 * EGG_1 + a2 * EGG2 + a3 * EGG3 \qquad (3\text{-}28)$$

where SMOLT and EGG_i are defined above, and a_i are scalar coefficients with units of number of smolts per egg, with $a1 > a_2 > a_3$, and $a_i \leq 1.0$. This equation defines a curve similar to that shown in Figure 3.3, with two intermediate inflection points. In the context of the least-cost model, each EGG_i in Eq. (3-28) is a state variable and the a_i are coefficients.

The next step is to account for the "consumption" of existing rearing habitat (at zero cost) and to allow for the addition of new rearing habitat (at a cost). Note that because the existing habitat can be used at zero cost it will be used first, that when new rearing habitat is added, each increment of additional habitat will "fill up" first with the most productive increment and later with the less productive increments, assuming that sufficient eggs are available.

In order to account for the use of rearing habitat, it is perhaps easiest to think of a certain amount of habitat required to raise each increment of eggs to smolts (the EGG_i in Eq. (3-28) above). This may be expressed mathematically as

$$HABUSE^j = b_1^J * EGG; + b_2^J * EGG; + b_3^J * EGG; \qquad (3\text{-}29)$$

where $HABUSE^J$ is the portion of the j^{th} unit of habitat that is used, and the b_i^J are coefficients relating egg rearing to the amount of habitat used, in units of habitat per egg. (Recall that 1 habitat unit is the quantity required to support 1,000 smolts at maximum carrying capacity.) The EGG_i are defined above. Note that in this case, $b_1 < b_2 < b_3$, since by assumption the "first" increment of eggs has a smaller marginal impact on habitat than the last.

There is, of course, an upper limit on the amount of existing rearing habitat that is available. This may be expressed mathematically as

$$HABUSE^{j} \leq 1.0 \tag{3-30}$$

and

$$\sum_{j} HABUSE^{j} \leq HABMAX \tag{3-31}$$

where HABMAX is the maximum number of habitat units at a particular rearing site, or the maximum carrying capacity of that site divided by 1,000. (By definition, this is the maximum carrying capacity for a particular stock).

The availability of habitat units can be increased by such activities as removing impediments to passage and increasing stream flows to increase the area available for rearing.

The mathematical relationships shown above can be extended to include new additions to the existing habitat. This is accomplished by adding the superscript "j" to the equations (as in Eqs. (3-29), (3-30), and (3-31) above) to denote the habitat increment being referenced. Three equations are required to describe smolt production of a particular stock. These three equations are egg production, the quantity of rearing habitat available, and smolt production. The first equation describes egg production. It was developed above in the section on egg production.

$$EGGS = c * ADS \tag{3-25}$$

where EGGS is the total number of eggs produced by a given stock, ADS is the number of returning adult spawners, and c is a stock-specific constant expressed in number of eggs per spawner.

The remaining equations concern rearing of fry to smolts. The next equation defines the egg productivity increments in habitat unit j:

$$EGGS^j = \sum_i EGG_i^j \quad (3\text{-}32)$$

where $EGGS^j$ is the number of eggs required to produce 1,000 smolts (at maximum carrying capacity) in habitat unit j, and EGG_i^j is the number of eggs in productivity increment i and habitat unit j. The productivity increment refers to the crowding effect noted above, and the habitat unit designates which "unit" of habitat is being used for production.

The next equation describes the number of smolts produced in habitat increment j.

$$SMOLT^j = \sum_i a_i^j * EGG_i^j \quad (3\text{-}33)$$

where $SMOLT^j$ is number of smolts produced in habitat unit j and a_i^j are coefficients expressed in number of smolts per egg. The EGG_i^j are defined above.

The next equation is used to describe the portion of the jth habitat unit used to raise smolts.

$$HABUSE^j = \sum_i b_i^j * EGG_i^j \quad (3\text{-}34)$$

where $HABUSE^j$ is the portion of the jth habitat unit used to raise eggs to smolts and b_i^j are coefficients relating habitat use to the number of eggs. This equation is identical to Eq. (3-29).

The next equation provides limits on the variable $HABUSE^j$.

$$HABUSE^j \leq 1.0 \quad (3\text{-}35)$$

The following equation ensures that the number of habitat units used does not exceed the total available.

$$\sum_j HABUSE^j \leq HABMAX \qquad (3\text{-}36)$$

This equation is identical to Eq. (3-31) above.

As noted above, there are two principal components of habitat, stream flow and quantity of substrate. Continuing with the notation used above, this may be expressed as follows:

$$HABMAX = \sum_j f\,(Substrate_j, Flow_j) \qquad (3\text{-}37)$$

The following set of equations show how this can be formulated for inclusion into a least-cost model:

$$HABMAX_j = \min\,(SUB_j\,,\ c_j * Q_j) \qquad (3\text{-}38)$$

where SUB_j = Substrate, habitat unit j
Q_j = Flow (in cfs), habitat unit j
c_j = coefficient in units of habitat per unit (cfs) of flow

Either the substrate or the stream flow can be limiting in particular situations. Both are management alternatives potentially available to resource managers.

Inputs. This module has two principal inputs. They are returning adult spawners and stream flows at the various rearing sites. The returning adults are "produced" in the upstream migration of adults module described below. The stream flows are of two types. One type (of stream flow) is exogenous to the least-cost model. These flows occur on the unregulated tributaries and are estimated from historic streamflow data provided by the U.S. Geological Survey. The other type of stream flow is endogenous to the least-cost model and is an output of the hydrosystem modules described above. The provision of new habitat through increases in "substrate" (natural spawning areas) and through the construction of new and expanded hatcheries is included in this module. It is discussed in more detail below.

Outputs. There are three principal outputs of this module. They are smolts ready for input into the smolt migration module (described below), new and expanded natural rearing habitat, and new and expanded hatcheries.

Constraints. The only constraints in the smolt production module are the total amounts of habitat that are potentially available to each stock, including possible new additions that can be "installed" by the model. For the natural rearing areas, this is related to the potential amount of substrate available after removing obstructions, improving streambeds, and providing other improvements. For hatcheries, the maximum total available habitat is limited by the available water and by suitable expansion capacity.

Mathematical Relationships. The mathematical relationships for egg production and smolt production were developed in the introduction to the smolt production module. This section extends the earlier development and delineates the constraints in this part of the model. Some of the mathematical relationships developed earlier are re-used here without a full explanation. The equations developed below are primarily mass-balance relationships. The notation used in this section is generally consistent with that used in the introduction to this module rather than the notation scheme presented in Table 3.1. Because the development in this section pertains to particular stocks, for convenience the subscripts "i, t ,s" in Table 3.1 are not used.

The first set of constraints ensures that the eggs used in producing smolts do not exceed the eggs produced by adult spawners.

$$\sum_{j}^{M} \sum_{i}^{N} EGGUSE^{j} \leq c*ADS$$

or

$$\sum_{j}^{M} \sum_{i}^{N} EGGUSE_{i}^{j} - c*ADS \leq 0 \quad (3\text{-}39)$$

where $EGGUSE_{i}^{j}$ denotes the "use" of eggs to produce smolts, habitat unit j , productivity increment i, in number of eggs; ADS is the number of returning adult spawners of a particular stock; c is a coefficient for egg production by spawning adults, in number of eggs produced per adult (See Eq. 3-25); N is the number of productivity increments (See Eq. 3-32); and M is the number of habitat units.

A similar equation is used to describe the production of smolts in the jth habitat unit.

$$SMOLT_i^j = a_i^j * EGG_i^j$$

or in linear programming format

$$SMOLT_i^j - a_i^j * EGG_i^j = 0 \tag{3-40}$$

where $SMOLT_i^j$ represents the number of smolts, habitat unit j, productivity increment i, and a_i^j are coefficients for the number of smolts produced per egg, productivity increment i, habitat unit j.

Since both eggs and smolts are assumed to have the same characteristics regardless of the productivity increment or habitat unit that produces them, the smolts can be added together to produce a total, the input needed for the smolt migration module. This may be expressed as

$$SMOLTOT = \sum_j^M \sum_i^N SMOLT_i^j$$

or in linear programming format

$$\sum_j^M \sum_i^N SMOLT_i^j - SMOLTOT = 0 \tag{3-41}$$

where SMOLTOT is the total number of smolts of a particular stock available to migrate downstream.

The next set of constraints pertain to the use of rearing habitat in the production of smolts. They are all mass-balance relationships.

The first set of relationships ensures that the available habitat will not be exceeded by the requirements for rearing. The jth unit of habitat may be expressed, as before, as

$$HABUSE^j = \sum_i^N b_i^j * EGG_i^j \qquad (3\text{-}34)$$

where $HABUSE^j$ is the portion of the jth unit of habitat that is used, and b_i^j are coefficients relating egg rearing to the amount of habitat used, in units of habitat per egg, and where $HABUSE^j \leq 1.0$ from Eq. (3-30).

From Eq. (3-36),

$$\sum_j HABUSE^j \leq HABMAX$$

and from Eq. (3-34),

$$\sum_i^N b_i^j * EGG_i^j \leq HABMAX \qquad (3\text{-}42)$$

where HABMAX is the total rearing habitat available to a particular stock.

Since stream flow is one of the limiting factors for rearing habitat, the next equation describes the effects of stream flow on the availability of rearing habitat.

$$HABMAXQ_t = \sum_i a_i * QT_{i,t} \qquad (3\text{-}43)$$

where $HABMAXQ_t$ is the total habitat available to a particular stock (limited by streamflow rather than substrate) in time period t, $QT_{i,t}$ is the total flow at location i in time period t (in CFS), and a_i are coefficients in units of habitat per unit of stream flow. As shown in Eq. (3-43), the total habitat used for rearing in time period t, $\sum_j HABUSE^j$, cannot exceed the total habitat available in time period t, $HABMAXQ_t$, or

$$\sum_j HABUSE^j \leq HABMAXQ_t \tag{3-44}$$

The available habitat is also limited by substrate which includes a number of factors such as the available food supply and cover. The next equation ensures that the habitat used for rearing will not exceed the maximum available habitat limited by substrate.

$$\sum_i HABUSE^j \leq HABMAXSUB \tag{3-45}$$

where $HABUSE^j$ is defined above and HABMAXSUB is the total habitat available to a particular stock (limited by substrate rather than by streamflow) in time period t. Note that because of the wide variety of activities that can produce substrate, the HABMAXSUB activities can take on many forms depending on the specific situation in a particular area. This expression could be disaggregated if sufficient data were available (as it might be at hatcheries) into food, cover, area of stream bed for rearing, and so on.

Finally, as noted above, there may be a physical upper limit on the amount of rearing habitat potentially available to a given stock.

$$HABMAXQ \leq HABCAP$$

$$HABMAXSUB \leq HABCAP \tag{3-46}$$

where HABCAP is an exogenous physical upper limit on the quantity of habitat that can be produced.

There are costs associated with the addition of nev habitat as well as with the operation and maintenance of existing habitat. The costs of new habitat may be expressed as

$$\sum_{j}^{M} C_j * HABUSE^j \qquad (3\text{-}47)$$

where C_j is the unit cost (capital, operating, and maintenance) of providing and maintaining habitat unit j, and M is the total number of habitat units available to a particular stock. Note that existing habitat has no associated capital costs.

It may be necessary to define some of the habitat units, $HABUSE^j$, as integer variables to accommodate possible scale economies or to add relatively large amounts of habitat in discrete increments, as in removing passage obstructions to large spawning and rearing areas.

The rearing of fry to smolts often takes more than one month. If this is the case, it will be necessary to provide relationships for the "carry-over" from one month to the next. This is a straight-forward extension of the mathematical development above.

Costs. The costs in this module include the costs incurred in maintaining existing rearing habitat and in adding and maintaining new rearing habitat, including hatcheries. Other costs include the "opportunity costs" involved in changing instream flow patterns to accommodate the rearing of smolts.

Assumptions Required for Linear Program. The assumptions made to enable the smolt production module to be developed as a mixed-integer LP model are the following:

- Linear relationship between the number of eggs produced and the number of adult spawners.
- Declining marginal productivity of habitat as the number of smolts per unit of rearing habitat is increased.

If it develops that spawning habitat is also a limiting factor (in addition to rearing habitat), it will be necessary to develop a relationship between eggs and adult spawners similar to that for smolts and eggs.

Smolt Migration Module

The smolt migration module transports the smolts produced in the smolt production module to the ocean, accounting for mortality in transit.

Smolts face three kinds of obstacles along their journey to the ocean. The first is the reservoirs they pass through along the mainstem Columbia and Snake Rivers. The second is the dams they pass through or around. The third is the irrigation diversions, or irrigation withdrawal sites. Irrigation diversions have two kinds of effects on smolts. First, smolts can get drawn into the irrigation intakes. Second, withdrawals for irrigation result in reduced stream flows. These three obstacles are discussed in this module. As with the other modules, the focus for expository purposes is on a single stock, a single time period, and a single location.

Inputs. The smolt migration module requires four principal inputs, three from the hydrosystem modules and one from the fish production and migration modules. The first input is the total monthly flow entering each reservoir. The second input is the amount of water at each dam that is diverted through the turbines and the amount that is diverted over the spillway. The third input is the amount of water withdrawn for irrigation at each withdrawal site along the migration route. The fourth input is the number of smolts (of each stock) from the smolt production module.

outputs. There are two principal outputs of this module. The first output is the total number of smolts entering the estuary. This total is comprised of those smolts arriving “naturally” under their own power and those transported by barge or truck after being collected at upstream collection and diversion structures. The second output is the installation of new collection and diversion structures, and the use of barges or trucks. or both, to transport smolts downstream.

Constraints. There are two types of constraints in the module. The first type of constraint is mass balances. The second type of constraint is upper bounds on the efficiency of travelling screen and other collection apparatus used to collect smolts that have started into the turbines.

Mathematical Relationships. The mathematical relationships in the module will be developed in the order presented above--passage through reservoirs, passage through and around dams (and subsequent transport by truck or barge), and passage past irrigation withdrawal sites. Irrigation water withdrawal is not treated in detail. The effect of irrigation water withdrawals on instream flows is handled in the hydrosystem modules. The effect of reduced instream flows due to irrigation water withdrawals on smolt passage is handled in this module.

The reservoir mortality portion of the smolt migration module is based on the fish simulation model developed by Lee (1987) in another part of this project. That model treats smolts as “particles” with a fixed stock-specific probability of dying within a given time period. The fish simulation model is used to estimate the average number of smolts surviving passage through each reservoir as a function of the management variables under the control of the least-cost model. Thus, the relationship between the number of smolts surviving passage and the time of passage through the reservoir is incorporated in the least-cost model. The mathematical relationships used to govern reservoir mortality are developed below.

The probability of surviving passage of a particular reservoir is a function of the natural mortality rate and the residence time in the reservoir. The relationship may be expressed in exponential form as

$$P = e^{-\rho t_F} \qquad (3\text{-}48)$$

where

P = average probability of surviving the reservoir.

ρ = natural mortality rate of the stock, per day.

t_F = average residence time of the stock in the reservoir, in days.

The natural mortality rate of the stock, p, is assumed to be a constant and to be known prior to the analysis.

The average residence time of the stock in the reservoir (in days) is unknown prior to the analysis. It may be computed from the length of the reservoir, L, and the average velocity of the smolts traversing the reservoir, V_F, using the following expression

$$t_F = \frac{L}{V_F} \tag{3-49}$$

where t_F is the average residence time of the stock in the reservoir in days, L is the length of the reservoir in feet, and V_F is the average velocity of the smolts in the reservoir in feet per day. The length of the reservoir is known prior to the analysis, but the average velocity of the smolts is not.

The velocity of the smolts in the reservoir is assumed to be proportional to the velocity of the water in the "channel" (within the reservoir) that the smolts choose to swim in in traversing the reservoir. In general, the velocity of the water in the reservoir is not uniform (the same at every point in the reservoir). Therefore, the velocity of the water in the particular channel that the smolts choose to swim in may not be known at the time of the analysis, particularly if flows (and thus average velocities) are allowed to change. However, the average velocity of the water in the reservoir can be calculated fairly easily, and the average velocity of the smolts in the reservoir can be related to that. This is expressed in the next equation.

$$V_F = a\ VW \tag{3-50}$$

where V_W is the average velocity of the water in the reservoir in feet per day, a is an empirical coefficient relating the average velocity of the smolts in the reservoir to the average velocity of the water- in the reservoir, and V_F is defined above.

There are two similar approaches to estimating the average velocity of the water in the reservoir. The first approach involves using the length of the reservoir, L, the volume of water stored in the reservoir, S, and the flow out of the reservoir, Q, to compute the velocity. All three independent variables are known at the time of the analysis. Using this approach, the average velocity of water in the reservoir may be developed from

$$V_W = \frac{L}{t_W} \tag{3-51}$$

where

$$t_W = \frac{S}{a} \tag{3-52}$$

From Eqs. (3-51) and (3-52), the average velocity of water in the reservoir may be expressed as

$$V_W = \frac{LQ}{S} \tag{3-53}$$

where

V_W = average velocity of water in the reservoir, in feet per day.

L = length of the reservoir, in feet.

t_W = average residence time of the water in the reservoir, in days.

S = volume of water in the reservoir, in cubic feet.

Q = total outflow from the reservoir, in cubic feet per day.

Eq. (3-53) represents the average velocity of water in the reservoir. Actual velocities may vary widely between the lower and upper layers of the reservoir, as noted above. Actual water velocities may have a substantially greater impact on the average smolt velocity than the average water velocity. This is an empirical issue that needs to be investigated as part of the fish simulation model development.

The second approach to estimating the average velocity of the water in the reservoir involves using the average cross-sectional area of the reservoir, A, and the flow through (or out of) the reservoir, Q. The difficulty with this approach is it will be difficult to estimate the average cross-sectional area if it varies widely along the length of the reservoir. Using this approach, the average velocity of the water in the reservoir may be expressed as

$$V_W = \frac{Q}{A} \tag{3-54}$$

where

Q = total outflow from the reservoir, in cubic feet per day.

A = average cross-sectional area of the reservoir, in square feet.

The average cross-sectional area used to estimate the average water velocity expressed in Eq. (3-54) can either be the total cross-sectional area of the reservoir or the “effective” cross-sectional area.

The “effective” cross-sectional area is the area in a reservoir where the water is actually flowing. It will generally be somewhat less than the total cross-sectional area, since only a portion of the water in a large reservoir will actually be flowing with the remainder being more or less stagnant. As long as the “effective” cross-sectional area is known prior to the analysis, it can be allowed to vary with flow without affecting the ability to fit this functional form into a mixed-integer LP framework. Note that (3-53) and (3-54) are identical if one assumes that water velocity is the same throughout the reservoir.

The probability of surviving passage of a particular reservoir, expressed above as Eq. (3-48), requires an estimate of the average residence time of the smolts in the reservoir (in days). This residence time can be estimated by one of the methods described above or it can be estimated from the fish simulation model developed for this project by Lee (1987). Thus, the residence time of the smolts in the reservoirs may be expressed three ways. The first expression is

$$t_F = \frac{K_1}{Q} \tag{3-55}$$

where Q is the total outflow from the reservoir and K_1 is an empirical constant estimated directly from the fish simulation model developed by Lee (1987).

The second expression is based on measurements of the volume of water stored in the reservoir and the total outflow from the reservoir, Eq. (3-49) and (3-53), and the relationship between the average velocity of the smolts in the reservoir and the average velocity of the water in the reservoir, Eq. (3-50).

$$t_F = \frac{S}{aQ}$$

$$t_F = \frac{K_2}{Q} \tag{3-56}$$

where Q is the total outflow from the reservoir, S is the volume of water stored in the reservoir, a is an empirical coefficient defined by Eq. (3-50), and K_2 is an empirical constant computed by dividing S by a.

The third expression is based on measurements of the length of the reservoir, the total cross-sectional area (or “effective” cross-sectional area) of the reservoir, and the total outflow from the reservoir, Eqs. (3-49) and (3-54), and the relationship between the average velocity of the smolts in the reservoir and the average velocity of the water in the reservoir, Eq. (3-50).

$$t_F = \frac{LA}{aQ}$$

$$t_F = \frac{K_3}{Q} \tag{3-57}$$

where, as before, Q is the total outflow from the reservoir, L is the length of the reservoir, A is the cross-sectional area of the reservoir, a is an empirical coefficient defined by Eq. (3-50), and K_3 is an empirical constant computed as LA/a.

All three expressions for the average residence time of the smolts in the reservoir developed above, Eqs. (3-55), (3-56), and (3-57), are of the same form for purposes of inclusion in the least-cost model.

Since for the fixed-head reservoirs S, L, and A are all constants and a is empirically derived, and since ρ is assumed to be constant and known for a particular stock, the probability of surviving passage of a particular reservoir, Eq. (3-48), may be expressed as

$$P = e^{-K/Q} \qquad (3\text{-}58)$$

where $K=\rho K_1$, $K=\rho K_2$, or $K=\rho K_3$ depending on the approach taken to estimate the residence time of smolts in the reservoir, t.

Eq. (3-58) has both reasonable and desirable properties from the perspective of estimating the probability of survival. As Q increases, P approaches 1 in the limit of $Q=\infty$. As Q decreases, P approaches zero in the limit of Q=O.

The next step in the process of developing a least-cost model is to fit Eq. (3-58) into a mixed-integer LP framework. Although Eq. (3-58) has desirable properties, including declining marginal impacts of increasing flows, the function is nonlinear with respect to flow. This problem can be handled in the model by dividing the flow into N intervals and using integer variables. If irrigation water withdrawals along the reservoir result in significant differences between the flow in the upstream portion of the reservoir and the flow in the downstream portion of the reservoir (near the dam), it may be necessary to divide the reservoir into two or more transverse sections. If this is necessary, additional integer variables will be required.

The equations in this portion of the model may be expressed as follows:

$$TF_{i,t} = QMAX_1*Q_1^* + QMAX_2*Q_2^* + \ldots + QMAX_N*Q_N^* \qquad (3\text{-}59)$$

where $TF_{i,t}$ is the total flow through reservoir i in time period t and $QMAX_j$ are the "integer" flows through the reservoir.

Eq. (3-59) divides the total flow, $TF_{i,t}$, into N discrete intervals. Q_j^* are integer (0,1) variables. $QMAX_j$ are the quantities of total flow associated with each Q_j^*, with $QMAX_j \geq QMAX_{j-1}$. The $QMAX_j$ (measured in cubic feet per second) are constants. Eq. (3-59) ensures that all the flow in the reservoir ($TF_{i,t}$) is accounted for by the integer variable. Thus, if any integer variable, $Q_j^* = 1$, the total flow, $TF_{i,t}$, is equal to $QMAX_j$ and > $QMAX_{j-1}$.

The next step is to establish a constraint that ensures that only one of the Q_j^* takes on a non-zero value at a time.

$$\sum_{j}^{N} Q_j^* \leq 1 \qquad (3\text{-}60)$$

The different flows corresponding to the N intervals are used to account for the mortality of smolts in the reservoir. The first step is to create a series of N "capacity" activities that pass the smolts through the reservoir using the flow rates corresponding to the Q_j^* provided above.

$$SMIG_j = Q_j^* * SMAX \qquad j = 1,\ldots,N$$

or in linear programming format,

$$Q_j^* * SMAX - SMIG_j = 0 \qquad j = 1,\ldots,N \qquad (3\text{-}61)$$

where the Q_j^* are from Eq. (3-59), and SMAX is the maximum number of smolts expected to pass reservoir i in time period t. $SMIG_j$, defined below, are used to "pass" smolts along in varying proportions depending on the total flow through the reservoir.

$$SMOLT_{i,t} - \sum_{j}^{N} SMIG_j = 0 \tag{3-62}$$

Eq. (3-62) ensures that smolts passed by the $SMIG_j$ equal the incoming smolts (of a particular stock), $SMOLT_{i,t}$.

The next step is to account for mortality in the reservoir (or in the different sections of the reservoir) which depends on the average residence time of the smolts in the reservoir, and thus on the flow (See Eq. (3-58)). For those reservoirs that are divided into two or more transverse sections for purposes of analysis, new notation is required. The upper most section is designated by the subscript i, the next downstream section by the subscript i+1, and so on.

The next expression is a mass balance equation that ensures that the number of smolts entering the $(i-1)^{th}$ section of the reservoir is equal to the number of smolts that survive the ith section (of the reservoir).

$$\sum_{i}^{N} SMIG_j * b_j \quad SMOLT_{i+1\ t} = 0 \tag{3-63}$$

where the b_j are defined as

$$b_j = e^{-(k/QMAXj)} \tag{3-64}$$

and as such are the proportion of smolts surviving passage of the reservoir. The "k" term is both stock and reservoir specific. as shown above in Eqs. (3-55), (3-56), and (3-57), and the $QMAX_j$ approximates the total flow through the reservoir. Note that due to the integer constraints on the Q_j^*, only one $SMIG_j$ actually operates for each reservoir and time period. Note also that and where

$$QT_{i,t} = QS_{i,t} + QG_{i,t} \tag{3-67}$$

Note from Eqs. (3-65) and (3-66) that $PSP_i + PTU_i = 1$. Thus, the number of smolts passing diversion screens or through the turbines at dam i may be expressed as

$$SMOLTT_i = PTU_i * SMOLT_i \tag{3-68}$$

and the number of smolts passing over the spillway at dam i may be expressed as

$$SMOLTS_i = PSP_i * SMOLT_i \tag{3-69}$$

where

$SMOLTT_i$ = number of smolts passing the turbines at dam i

$SMOLTS_i$ = number of smolts passing the spillway at dam i

$SMOLT_i$ = total number of smolts arriving at dam i.

While this is mathematically straightforward, Eqs. (3-68) and (3-69) are both nonlinear because of the proportion terms, PTU_i and PSP_i, and therefore they cannot be incorporated in a linear programming framework. Moreover, they are difficult to incorporate in a mixed-integer LP framework. The reason for the difficulty is that the expressions for the proportions of flow passing through the turbines and over the spillway, PSP and PTU, are both nonlinear functions of two decision variables, as shown in Eqs. (3-65) and (3-66) above. Fortunately, however, it is possible to re-use the integer variables created above for reservoir passage to approximate these proportions.

The first step is to calculate the proportion of water that goes through the turbines. The total flow through the turbines at dam i may be expressed as the sum of the integer flows.

and where

$$QT_{i,t} = QS_{i,t} + QG_{i,t} \tag{3-67}$$

Note from Eqs. (3-65) and (3-66) that $PSP_i + PTU_i = 1$. Thus, the number of smolts passing diversion screens or through the turbines at dam i may be expressed as

$$SMOLTT_i = PTU_i * SMOLT_i \tag{3-68}$$

and the number of smolts passing over the spillway at dam i may be expressed as

$$SMOLTS_i = PSP_i * SMOLT_i \tag{3-69}$$

where

$SMOLTT_i$ = number of smolts passing the turbines at dam i

$SMOLTS_i$ = number of smolts passing the spillway at dam i

$SMOLT_i$ = total number of smolts arriving at dam i.

While this is mathematically straightforward, Eqs. (3-68) and (3-69) are both nonlinear because of the proportion terms, PTU_i and PSP_i, and therefore they cannot be incorporated in a linear programming framework. Moreover, they are difficult to incorporate in a mixed-integer LP framework.. The reason for the difficulty is that the expressions for the proportions of flow passing through the turbines and over the spillway, PSP and PTU, are both nonlinear functions of two decision variables, as shown in Eqs. (3-65) and (3-66) above. Fortunately, however, it is possible to re-use the integer variables created above for reservoir passage to approximate these proportions.

The first step is to calculate the proportion of water that goes through the turbines. The total flow through the turbines at dam i may be expressed as the sum of the integer flows.

$$QG_i = \sum_j^N Q_j^* * QGE_j \qquad (3\text{-}70)$$

where QG_i is the total flow through the turbines at dam i (in CFS), QGE_j represents a set of N flows through the turbines! also in CFS, and Q_j^* are integer (0. 1) variables. In the development that follows, the "i" subscript is dropped for exposi tory convenience. The purpose of Eq. (3-70) is to divide the flow through the tur bines into N discrete intervals for subsequent use in cal ulating the pr opor tions of total flow that go through the turbines and that pass over the spillway.

The next step is to calculate the proportion of total flow that passes ***over*** the spillway. This may be calculated from the proportion that passes through the turbi nes. as the sum of these two proportions is equal to 1.0. Let $PERSP_j$ be the proportion of total flow that passes over the spillway corresponding to the proportion of total flow that passes through the turbines, $(QGE_j/QMAX_j)$. The mass balance for these proportions may be expressed as

$$Q_j^* * PERSP_j + Q_j^* * (QGE_j/QMAX_j) = Q_j^*(1.0) \qquad j=1,\ldots,N \qquad (3\text{-}71)$$

For $Q_j^* = 1$, Eq. (3-71) reduces to

$$PERSP_j + (QGE_j/QMAX_j) = 1.0$$

or in linear programming format,

$$PERSP_j + (QGE_j/QMAX_j) - 1.0 = 0 \qquad (3\text{-}72)$$

where Q_j^* are the integer (0,1) variables discussed above in the reservoir mortality section, QGE are the flows through the turbines defined above, and $QMAX_j$ are the total flows through the reservoir for each discrete interval of flow (also used in the reservoir mortality section). $QMAX_j$ are constants in the analysis. $PERSP_j$ are the proportions of water going over the spillway for the N intervals of flow through the turbines.

The set of equations expressed above as Eqs. (3-71) and (3-72) requires an explanation since its meaning is not intuitively obvious and these equations are essential for driving the equations that follow. The first thing to note from this equation set is that of the N equations only one will have a non-zero value for its corresponding $PERSP_j$ for any given time period. The reason for this is that only one of the Q_j^* will have a non-zero value, due to the constraints placed upon them in the reservoir mortality section. (See Eq. (3-60)). Since only one Q_j^* will be non-zero, only one pair of QGE_j and $PERSP_j$ can be non-zero without violating Eq. (3-71). This is shown as Eq. (3-72).

The second thing to note from Eq. (3-71) is that since the values of $QMAX_j$ are known prior to the analysis, none of the equations are nonlinear with respect to the three decision variables, Q_j^*, QGE_j, and $PERSP_j$. Note also that the $QMAX_j$ corresponding to the Q_j^* that is operating at a non-zero value is approximately equal to the total flow past the dam, QT. Thus, the objective of defining the proportion of water going over the spillway, $PERSP_j$ as a strictly linear function of other decision variables has been achieved.

The next step is to use the $PERSP_j$ calculated above to "guide" the smolts in proportion to the two principal flows passing the dam. This is done in the equations developed below. These equations are shown for a particular stock of migrating smolts. They could be extended to multiple stocks passing the same dam in the same time period if it were desirable to do so.

The first step is to sum the $PERSP_j$ (only one of which will be non-zero). This will be used to calculate the proportion of smolts passing over the spillway.

$$\sum_{j}^{N} PERSP_j - \sum_{k}^{M} PERFORSM_k * b_k \geq 0 \quad . \qquad k=1,\ldots,M \tag{3-73}$$

$$\sum_{k}^{M} PERFORSM_k \leq 1 \tag{3-74}$$

where $PERSP_j$ is defined above, $PERFORSM_k$ are integer (0,1) variables, and bk is a set of coefficients in the range of 0.0, 0.1,0.2,....,1.0. Eq. (3-74) is constrained to be equal to or less than 1 so that no more than one of the $PERFORSM_k$ will take on a non-zero value.

As with equation set (3-71) above, equation sets (3-73) and (3-74) also require an explanation. The first term on the left side of Eq. (3-73), Σ PERSP;, is the total proportion of water spilled. The second term on the left side, $\Sigma\, PERFORSM_k$, is more complicated. $PERFORSM_k$, k=1,...,M, are integer (0,1) variables, only one of which can be non-zero in a particular time period. Their purpose, in combination with b_k, is to calculate an approximate integer equivalency of the total proportion of water going over the spillway. This proportion is used subsequently to "guide" smolts over the spillway and through the turbine intakes in the proper proportions as the water. Thus, given a range for b_k (from 0.0 to 1.0, incremented by 0.1), the term $\Sigma\, PERFORSM_k * b_k$ in Eq. (3-73) approximates the exact spill proportion calculated in the $\Sigma\, PERSP_J$.. In particular, b_k for the single non-zero $PERFORSM_k$ is both the approximate proportion of water going over the spillway and the approximate proportion of smolts going over the spillway. This is used below to move the smolts in accordance with the assumption that they move in proportion to the flows.

Once a b_k has been chosen that approximates the proportion of water spilled, the remainder of the problem is essentially one of accounting for the fate of smolts via a series of mass-balance equations. The equations that follow are for a single stock. The first step is to define another set of M variables, c_k. Let

$$c_k = 1 - b_k \qquad k=1,..,M \tag{3-75}$$

where c_k is the proportion of smolts heading toward the turbines, since b_k is the proportion going over the spillway. The set of c_k is calculated prior to the analysis. They are used below in the "routing" equations.

The next step is to account for the routes taken by the smolts through (and around) the dam. The next equation is a mass-balance equation that ensures that all the smolts of a particular stock are accounted for. It divides the smolts arriving at the dam into M intervals so that they can be "channelled" through the appropriate set of migration activities. In the development below, the "i, t, s" subscripts are dropped for expository convenience.

$$SMOLT_{i,t} = \sum_{k}^{M} SM_{i,t,k}$$

or in linear programming format,

$$SMOLT_{i,t} - \sum_{k}^{M} SM_{i,t,k} = 0 \tag{3-76}$$

where

$SMOLT_{i,t}$ = the number of smolts passing dam i in time period t. (These are the smolts from the reservoir migration section described above.)

$SM_{i,t,k}$ = the number of smolts at dam i in time period t in interval k.

The coefficients b_k and c_k are used to calculate the proportions of smolts taking the different routes through and around the dam. Since only one set of "k" activities can operate at any one time, this may be expressed as follows.

$$(PERFORSM_k * SMAX) - SM_k \geq 0 \qquad k=1,\ldots,M \tag{3-77}$$

where $PERFORSM_k$ are integer (0,1) variables, only one of which will be operating at a non-zero value (See Eq. 3-73), SMAX is the maximum number of smolts expected to pass the dam (set to a very large number), and SM_k are the number of smolts in the k^{th} integer interval passing the dam.

The purpose of Eq. (3-77) is to ensure that the correct proportions of smolts are chosen by the model to pass the spillway and the screens and turbines. This equation set guarantees that the smolts will use only the equation where $PERFORSM_k$ is equal to one.

The next step is to move smolts over the spillway. The following set of mass balance equations describes the passage of smolts over the spillway.

$$SM_k * b_k = SMSPILL_k \qquad k=1,....,M$$

or in linear programming format,

$$SM_k * b_k - SMSPILL_k = 0 \qquad k=1,....,M \qquad (3\text{-}78)$$

Equation set (3-78) ensures that the number of smolts passing over the spillway, $SMSPILL_k$, equals the total number of smolts approaching the dam, SM_k, multiplied by the proportion of water spilled, b_k.

The next set of mass balance equations describes the passage of smolts through the turbines and diversion equipment.

$$SM_k * c_k = SMTURB_k \qquad k=1,....,M$$

OK in linear programming formatt,

$$SM_k * c_k - SMTURB_k = 0 \qquad k=1,....,M \qquad (3\text{-}79)$$

This equation set is similar to Eq. (3-78) and ensures that the number of smolts entering the intake of the turbines, $SMTURB_k$, equals the total number of smolts arriving at the dam, SM_k, multiplied by the proportion of water passing through the turbines, c_k.

The next two equations “regroup” the smolts that pass over the spillway and through the turbines via the integer intervals into single quantities (numbers of smolts), one for the total number of smolts passing over the spillway and the other for the total number of smolts passing through the turbines. This is done for ease in accounting for mortality and other management options. The first equation "regroups" the number of smolts passing over the spillway.

$$\sum_{k}^{M} SMSPILL_k = SMSPILLT$$

or in linear programming format

$$\sum_{k}^{M} SMSPILL_k - SMSPILLT = 0 \qquad (3\text{-}80)$$

where SMSPILLT is the total number of smolts passing the spillway. The next equation "regroups" the number of smolts passing through the turbines.

$$\sum_{k}^{M} SMTURB_k = SMTURBT$$

or in linear programming format,

$$\sum_{k}^{M} SMTURB_k - SMTURBT = 0 \qquad (3\text{-}81)$$

where SMTURBT is the total number of smolts passing through the turbines.

The next step is to develop a relationship that describes the number of smolts that survive passage of the dam. For this, it is necessary to distinguish between two types of dams. Some dams have facilities to assist the smolts to pass it; others do not. For dams with no bypass or collection facilities, projecting survival is a matter of accounting for spillway and turbine mortality and grouping the survivors into a single variable after they have arrived below the dam. This is expressed in the following mass balance equation.

$$SMOLT_{i+1,t} = SMSPILLT * d + SMTURBT * e$$

or in linear programming format,

$$SMSPILLT * d + SMTURBT * e - SMOLT_{i+1,t} = 0 \tag{3-82}$$

where SMSPILLT is the total number of smolts passing over the spillway, "d" is the proportion of smolts surviving, SMTURBT is the total number of smolts passing through the turbines, "e" is the proportion of those surviving, and $SMOLT_{i+1,t}$ is the total number of smolts in time period t surviving the dam.

Eq. (3-82) is adequate if there are no options in the model for collecting the smolts that enter the turbine intakes and either diverting them around the dam or collecting them and transporting them via barge or truck to the estuary below Bonneville Dam.

If there are smolt collection and bypass facilities at the dam, Eq. (3-82) needs to be expanded to include those facilities. This can be achieved by adding a smolt collection activity, SMCOLLT, to the mass balance equation.

$$SMOLT_{i+1,t} = SMSPILLT * d + (l-g)* SMTURBT*e + SMCOLLT *f \tag{3-83}$$

where SMCOLLT is the number of smolts collected on the screen at the approach to the turbines and diverted around the dam, "g" is the collection efficiency of the screens, and "f" is the proportion of those surviving, and where

$$SMCOLLT = g*SMTURBT \tag{3-84}$$

For use in the model, it is more convenient to express Eqs. (3-83) and (3-84) in linear programming format as

$$SMSPILLT*d + (l-g) * SMTURBT*e + SMCOLLT*f - SMOLT_{i+1,t} = 0 \tag{3-85}$$

$$g * SMTURBT - SMCOLLT = 0 \tag{3-86}$$

where $SMOLT_{i+1,t}$ is the number of smolts that survive passage of dam i. Installation and operation of diversion equipment is straight-forward, and is so is not covered here.

Costs. The only costs in this module are those associated with the installation and operation of smolt collection and bypass facilities. There may also be “opportunity costs” associated with the hydropower and irrigation diversions foregone in providing more water to assist smolts around dams and on their downstream journey to the ocean, but these are reflected in the hydropower and irrigation modules.

Assumptions Required for Linear Program. Smolt migration cannot be incorporated directly into an LP format without using integer variables. This requires making the assumption that approximating the behavior of smolts with a reasonable number of integer variables will be accurate enough to meet the goals of the least-cost modelling effort. See Chapter 4 for a discussion of these issues.

Ocean Harvest and Survival Module

The ocean harvest and survival module is based on the ocean harvest model developed by Norton (1987). This model is discussed in detail in a separate part (see part III). The ocean harvest model is designed to identify the level of catch that maximizes economic welfare (measured as the sum of the producer and consumer surplus). It is tentatively structured as a nonlinear programming model and receives as input smolts and produces as output the level of catch and the number of adults that escape the ocean fishery. There are several empirical issues to be resolved before the ocean harvest model is actually constructed, and it may well turn out to be a comparatively simple linear program.

The least-cost model described in this chapter will incorporate the ocean harvest model in one of two ways, depending on the eventual structure of the ocean harvest model. If the ocean harvest model is structured as a linear programming model, it will be incorporated directly into the matrix of the least-cost model. If the ocean harvest model is structured as a nonlinear programming model, a response-surface mapping approach will be used. The inputs and outputs of this module are the same regardless of the structure of the ocean harvest model.

Inputs. The input to this module is the number of smolts arriving at the estuary below Bonneville Dam, disaggregated by stock and by year of arrival. There are no other inputs.

outputs. The outputs from this model include the level of the commercial ocean harvest (or catch) and the number of adults that escape the ocean fishery and return to the estuary below Bonneville Dam, disaggregated by stock and by month of arrival.

Mathematical Relationships. The main activities in the ocean harvest module are the relationships between smolts entering the ocean, adults in the ocean (by age class), natural mortality, level of ocean harvest, and the adults that escape the ocean fishery. The tentative management objective as the ocean harvest model is currently structured is to maximize consumer plus producer surplus subject to returning enough adults to the estuary to satisfy the needs both for in-river harvest and for spawning. The particular ocean harvest methods selected by the model, the level of natural mortality, and the number of adults surviving the ocean fishery will be provided by the ocean harvest model described by Norton in part III.

If the ocean harvest model is structured as a linear programming model, it will be relatively straightforward to incorporate it directly in the least-cost model as the ocean harvest and survival module. This module would be “supplied” with smolts from the smolt migration model (discussed above) and it would “return” adults to the upstream migration of adults module (discussed below). The mathematical relationships for this are developed below, for a particular stock and age class.

The number of adults escaping the ocean fishery and returning to the Columbia River to spawn may be expressed as

$$ADR = f\,(SMOLTS,\ Harvest\ Policy) \qquad (3\text{-}87)$$

where

ADR = number of adults returning to the estuary below Bonneville Dam.
SMOLTS = number of smolts entering the estuary below Bonneville Dam.
Harvest Policy = exogenously defined harvest policy.

The details of this relationship are included in the ocean harvest model presented in part III. If it is linear, it can be incorporated directly in the LP tableau of the least-cost model. If it is nonlinear, it cannot be incorporated directly into the least-cost model. A response-surface mapping approach will be required.

Costs. There are two types of costs in this module. One type is the direct resource costs of ocean harvest due to possible restrictions intended to protect the fishery. The other type of costs is the "opportunity costs" of harvest restrictions.

Assumptions Required for Linear Program. If the ocean harvest model is structured as a linear programming model, no assumption will be necessary to incorporate it in the least-cost model, beyond those described in part III. If the ocean model is nonlinear, the generated response-surface must be close enough to the actual response surface to obtain reasonably accurate results in the least-cost model.

Upstream Migration of Adults Module

This module takes adults escaping the ocean fishery and returning to the estuary below Bonneville Dam, and follows them in their upstream migration to their spawning areas, accounting for both in-river harvest and mortality along their migration routes. As with most of the other modules, there are a number of management alternatives in this module including improvements in upstream passage and the level of in-river harvest. Most of the equations in this module are mass-balances, since the module is primarily an "accounting" system to track the upstream migration of adults.

Inputs. The principal input to this module is the number of adults returning to the Columbia River below Bonneville Dam, disaggregated by stock and month of return.

Outputs. The principal outputs from the module are the number of adults harvested in-river and the number of adult spawners returning to either natural spawning grounds or hatcheries. A secondary output is newly installed capital equipment to aid in upstream passage of the adults.

Mathematical Relationships. There are two classes of activities in this module. One class concerns upstream mortality (and reducing it where it is both feasible and desirable). The other class involves in-river harvest. No special programming is required for this. In general, the number of adults (of a particular stock) is a function of the number of adults at the previous stage, such as a reservoir, dam, or reach of free-flowing water, minus losses due to harvest and to mortality at the dams and in the reservoirs. This may be expressed mathematically as follows.

$$ADR_{i,t,s} = ADR_{i-1,t,s} - ADH_{i,t,s} - ADM_{i,t,s} \qquad (3\text{-}88)$$

where $ADR_{i,t,s}$ represents the number of adults of species s at location i in time t, $ADH_{i,t,s}$ is the number of adults harvested, and $ADM_{i,t,s}$ is the number of adults dying due to causes other than harvest. In subsequent discussion, the "t,s" subscripts are dropped for expository convenience.

Equation (3-88) states that the number of adults is equal to the adults at the previous stage minus the number lost due to in-river harvest and to mortality. This is a mass-balance relationship whose purpose is to account for losses among the upstream migrating adults.

The next step is to calculate the number of adults harvested, ADH_i, and the number of adults that die due to other causes, ADM_i. ADH_i is a function of the adults potentially available for harvest, ADR_i, and the harvest techniques employed, which are assumed to be under the control of the least-cost model. It is assumed that the timing of upstream runs of different stocks is sufficiently diverse that a particular stock could be harvested without affecting other stocks, so that a mixed-stock fishery is not a problem.

The least-cost model can be operated in one of two ways. The first way is to constrain total harvest of a particular stock to be less than an exogenously defined maximum. The second way is to restrict the harvest rate or the efficiency of the harvest methods to be less than an exogenously assigned maximum rate. In both cases, a mass-balance constraint is required so that the number of fish harvested does not exceed the total number available at any given time. This may be expressed as follows.

$$ADH_i = a * ADR_l \tag{3-89}$$

where ADH_i is the number of adults harvested, ADR_i is the number of adults in the river, and "a" is a constant, $<$ 1, relating the number harvested to the total number of adults.

As already noted, a constraint can be placed on in-river harvest.

$$ADH_i \leq ADHMAX_i \tag{3-90}$$

where $ADHMAX_i$ is the maximum permissible harvest at location i.

A similar constraint could be placed on the total harvest of a particular stock or on the total harvest of a combination of stocks. The equations for these combination constraints are a straightforward extension of Eq. (3-90). They are not shown here. Restricting the harvest methods used can be done by exogenously varying the values of "a" in Eq. (3-89). Since the" a" in Eq. (3-89) is $<$ 1, mass-balance is ensured.

Mortality is handled in a similar fashion. Present understanding is that dam mortality is the primary adult mortality problem in the basin. If reservoir mortality also proves to be a problem, the approach shown below can be extended to encompass that as well.

There are several assumptions that underlie estimates of dam mortality in this module. First, the probability of successful passage at each dam is independent of the probability of passage at other dams. Second, reproductive success of the fish that pass the dams is independent of the management alternatives vis a vis upstream passage. Third, the probability of successful passage is independent of the harvest methods. Finally, dams that presently block upstream passage altogether will not be modified to permit passage. Given these assumptions, dam mortality and the management alternatives to reduce it can be handled quite easily in the least-cost model.

Since the probability of successful passage is assumed to be independent of other dams, this can be expressed for a "typical" dam as

$$ADM_i = ADR_{i-1} * b \qquad (3\text{-}91)$$

where ADM_i is the number of adults that die at dam i, ADR_{i-1} is the number of adults arriving just below dam i, and "b" is the probability of dying while attempting to pass dam i.

Equation (3-91) encompasses the assumptions noted above. If it is technically possible to improve dam passage (at some cost), this can be incorporated in the least-cost model by allowing alternative passage activities that reduce the "b" in Eq. (3-91) while adding costs to the objective function. For example,

$$ADM_i^* = ADR_{i-1} * b^* \qquad (3\text{-}92)$$

and

$$ADMCOST = ADR_{i-1} * c \qquad (3\text{-}93)$$

where ADM_i^* is the number of adults that die at dam i with a reduced mortality probability, b*. The cost of the activity, ADMCOST, is the product of the number of adults just below dam i, ADR_{i-1}, and the unit cost of the passage activity, c.

Equation (3-93) assumes that there are no scale economies in activities that improve passage around dams. If there are scale economies, or if there are significant fixed costs in these activities, it will be necessary to include integer variables to account the scale economies and fixed costs.

Costs. The only costs for this module are the opportunity costs of restricting in-river harvest, and the costs of improving passage facilities.

Assumptions Required for Linear Program. The assumptions required to include the upstream migration of adults in the LP matrix are described in the section on mathematical relationships above.

CONCLUDING COMMENTS

This concludes the description of the least-cost model. The purpose of this model, as described in the introduction, is to assist in identifying system-wide least cost fish mitigation alternatives. Such a model is needed because it is unlikely that the least-cost alternatives or set of alternatives can be identified using the hydrosystem and fish simulation models described briefly in chapter 1 and in more detail by Lee (1987). Nonetheless, this will not be achieved without considerable effort and without overcoming computational problems unique to the various "optimization" algorithms. These computational problems are due partly to model structure and partly to model size.

The approach taken to the development of the least-cost model, as described in the introduction to this chapter, has been, first, to identify the model structure and, second, to estimate its size. Both model structure and size are important factors in assessing the feasibility of applying the model in real situations.

The priority for model structure, based both on experience and on ease of application, has been ordinary linear programming (LP), mixed integer linear programming, and nonlinear programming, in that order. The reason for establishing this priority is that the computational problem get progressively more severe in moving from ordinary LP to nonlinear programming.

The mathematical development in this chapter has demonstrated that it will not be possible to construct a least-cost model for fish mitigation in the Columbia River Basin using ordinary linear programming (LP). However, at least in principle, it will be possible to construct a least-cost model using mixed integer linear programming. Moreover, it seems likely that such a model could be made sufficiently accurate to be helpful in identifying system-wide least cost fish mitigation strategies.

In the next chapter, the size of the model will be estimated. This will involve estimating the number of management and state variables (the columns in the LP tableau, or matrix), the number of relationships used to relate or constrain, or both, the management and state variables in the model), the number of non-zero elements in the LP tableau, the number of integer variables, and the number of constraining relationships used in conjunction with the integer variables. The resulting model size will be assessed in light of the available computer software and hardware.

Chapter 4

The Feasibility of Implementing the Least-Cost Model

INTRODUCTION

Chapter 3 demonstrated that the least-cost fish mitigation model will fit mathematically and conceptually into a mixed-integer linear programming framework. The conceptual model developed in there is extremely large, for two reasons. First, the number of stocks, reservoirs, dams, and mitigation alternatives is enormous, since the Columbia basin is a very large and diverse region. Second, it is likely to take several salmonid generations for the stocks in the region to double in size, the overall goal adopted by the Power Planning Council for system planning purposes.

The size of the proposed least-cost model poses two problems with respect to actually constructing and solving the large system of equations outlined in the previous chapter. The first concerns the data requirements for the model. The amount of information needed to construct the model is very large, and it seems likely that there will be difficulties with both finding the requisite primary data (on stock characteristics, mitigation alternatives, hatchery production, etc.) and generating the coefficients required to solve the least-cost model in any type of mathematical programming framework. Fortunately, the data required for the least-cost model is essentially the same as that needed for the system-wide simulation discussed in Chapter 2. The equations needed for the least-cost model will also be based upon those used for the simulation model. Therefore, data availability does not pose problems that are unique to this model. Chapter 6 addresses the practical problems of data sources for models that might be built in the future. A related issue is that the biological relationships are not known with a very high degree of certainty, which makes it more difficult to formulate mathematical models of fish behavior. As with the data availability, these problems are shared by both simulation and optimization. The stochastic nature of some of the phenomena associated with both the hydrosystem and the fisheries will need to be investigated, as these may present some problems for the least-cost model. Two possible ways to handle the stochastic problems are stochastic optimization or multiple, Monte Carlo runs of the least cost model with the exogenous, stochastic inputs, including hydrology and power loads, varied exogenously.

Even if the data needed were readily available in an easily usable form, and there were no uncertainty associated with the phenomena being modelled, however, other problems would remain. This category can be described as analytical problems. It includes the mathematical and computational feasibility of solving any large model where iterative solution methods are required to find a solution which minimizes (or maximizes) an objective function and simultaneously meet an exogenous set of constraints. The mathematical problems are caused primarily by the large size of the model. Round-off errors can occur in almost any non-integer calculation performed on a digital computer, but they are far more likely to be a problem with large models than with small ones. Serious round-off errors may result in solutions of a model that are inaccurate and misleading as a guide to policy analysis. In addition, since the computer time required to solve a model increases rapidly as the size of the model increases, at some point models become impossible to solve in any practical time period. This is especially true of mixed-integer linear problems, where the solution times increase as an exponential function of the number of integer variables in the model.

These analytical problems will almost certainly arise in the attempts to build and solve the model outlined in Chapter 3. The remainder of this chapter attempts to show the major causes of the problems, and suggests some possible ways to address the analytical issues involved. It ends by concluding that the analytical problems caused by the large size of the least-cost model may be solvable, using matrix decomposition techniques.

With the exception of the concluding section, the remainder of this chapter assumes some familiarity with operations research techniques, including linear programming, integer and mixed-integer programming, and the computational implications of increasing and decreasing model size. The reader who is not familiar with the concepts involved in the application of these methods to applied problems may therefore wish to skip to the conclusion. This can be done without loss of continuity, although some of the conclusions will need to be taken on faith.

MODEL SIZE

Whenever one encounters problems with models that are too large to solve at reasonable cost and within some reasonable time, there are three approaches take to the problem. The first is simply to give up the project as infeasible, an option that is not explored further in this chapter. The second is to reduce the size of the model by reducing the amount of information contained in it or, put differently, by restricting its scope and resolution. The third is to look at ways of structuring the problem that will give the same answer as would the original problem's solution, but that use different methods to arrive at that solution. In this discussion, we first examine the size of the model under different assumptions about its scope and resolution, and then explore a method for restructuring the problem to improve its computational feasibility without reducing the accuracy of the results.

A number of assumptions underlie the models' sizes that we present here. For both the hydrosystem and the biological sides of the model, we assume that all "integer" activities in Chapter 3 are broken up into 10 intervals. Within a model "year", we assume that there will be twelve monthly time periods.

For the hydro side, we assume 30 dams, of which 4 are variable-head hydropower, 19 are run-of-river hydropower, and 7 are storage-only with no significant generating capacity. These figures are based on a study of the various Corps of Engineers and Northwest Power Planning Council publications and exclude any dam having both less than 1 percent of the total storage capacity and less than 1 percent of the total generating capacity in the region. The are two obvious problems with this type of classification. The first is that it may ignore dams that are regionally insignificant but that are important on a sub-basin level. The second problem is that there may be small dams that have neither significant storage nor substantial generating capacity that cause serious problems for fish due to inadequate passage facilities. The second problem is not a severe one for estimating the model size, however, since the dams have only limited management alternatives associated with them. In any case, neither type of “small” dam will be picked up in this classification scheme. We further assume 20 major irrigation withdraw1 sites in the region.

For the fish, we assume fifty stocks, each of which passes an average of 6-8 downstream dams. Finally, we assume that the ocean harvest model will be very modest in size, at least in comparison to the freshwater portion of the least-cost model.

Having stated the various models to be considered (under the assumption that model structure remains as discussed in Chapter 3) and the assumptions that are behind each of the models, we describe them in Table 4.1. Each row in the table summarizes the size of a mixed-integer linear program (LP) formulated under different assumptions about the number of time periods to be modelled and whether or not to include both the hydrosystem and fish, or only one of the two. The first row represents the model developed in Chapter 3, while the other rows represent various ways to “subset” the model into smaller LP’ s.

The columns in Table 4.1 are defined below. The “Number of years” simply indicates the total years covered in each LP. “N” denotes the proportion of years that is included. If N=l, then every year is included; if N=3, then every third year is included. “Hydrology Included” indicates whether or not the hydrosystem (i.e., hydroelectricity, irrigation, and storage) is included, while “Biology Included” indicates whether or not the salmonid spawning, migration, and harvest activities are included. The next column indicates the number of constraints in the models, and the following column shows the number of integer variables required to formulate the model in a

Table 4.1.
Size of the Least-Cost Model Under Different Assumptions
About Model Structure

Number of years	"N"	Hydrology Included	Biology Included	Number Of Rows	Number of Integer Variables	Feasibility Estimate
15	1	Yes	Yes	500,000	137,000	
15	1	Yes	No	100,000	7,000	–
15	1	No	Yes	400,000	130,000	––
15	3	Yes	Yes	165,000	45,000	-
15	3	Yes	No	35,000	2,500	+
15	3	No	Yes	135,000	42,000	-
1	1	Yes	Yes	34,000	9,000	+
1	1	Yes	No	6,000	500	++
1	1	No	Yes	28,000	8,500	+

mixed-integer framework. Other parameters that describe the size of the constraint matrix, including the number of columns, the number of non-zero coefficients, and so forth are not included, since most published work on solution time and difficulty assert that the number of constraints and the number of integer variables are the most important parameters that influence computational performance. The model size estimates that are included are only rough approximations, accurate to at most two significant digits, and are based on the model formulation described in Chapter 3 and the assumptions about the dams, stocks, and irrigation sites noted above. The last column contains an estimate of the relative computational feasibility of the different models, ranging from extremely low ("--") to very high ("++").

As can be seen from the table, the models range in size from large to truly enormous. The largest is the model outlined in Chapter 3, which includes 15 years of monthly data on both the hydrosystem and the fish biology, while the smallest is the single-year model of the hydrosystem only. The reader will note that the feasibility estimates vary inversely with model size. The reason for this is that the rules of thumb within the operations research literature suggest that, for a simple continuous LP, the time required to solve the model is usually proportional to the cube of the number of constraint rows. Similar, although less well-developed, rules also exist for solution times required for mixed-integer models, based on the number of integer activities. Since the largest existing, continuous LP models that the authors are aware of have no more than 10,000 rows, the estimated feasibility of solving models with 500,000 rows is extremely low, especially when one considers the large number of integer variables in the largest model. On the other hand, several of the smaller models shown in Table 4.1 are quite probably feasible to solve both accurately and with reasonable solution times using standard LP packages and readily available computer systems.

The problem with the smaller models is of course relatively low scope and resolution. That is, the models that are comparatively easy to solve will give “correct” answers to only a small subset of the real problem. For example, in the table, the next-to-last line shows the number of rows and integer variables in a single-year hydrosystem-only model. This model, in our estimation, will probably be easy to solve with commercial LP/mixed integer LP software on a medium-size mainframe or fast minicomputer. Consider, however, the problem solved by that model. It answers the question of what the least-cost method to operate the hydrosystem is for a single year of operation, ignoring the fish entirely. Similarly, the last row of the table summarizes a model to optimize the management of the fisheries for a single year, ignoring the hydrosystem. Since the actual problem problem of interest includes management of both the hydrosystem and the fisheries over a fifteen-year planning horizon, these sub-sets of the largest model do not really answer the questions that are likely to be asked of them.

Given the conclusion that the large model that has the potential to answer questions about the least-cost methods for approximately doubling existing runs is not computationally feasible, we turn next to formal decomposition techniques that may allow the analyst to solve a model that does, in fact, give answers that are both analytically feasible and sufficiently detailed to serve as a guide for policy analysis.

DECOMPOSITION TECHNIQUES

Several methods have been developed by both theoretical and applied operations research workers that address the problem solving very large constrained optimization models. They are formally known as decomposition techniques. The techniques used in both hydrosystem modelling and electrical generation system planning are often based on a technique known as Bender decomposition. While this chapter will not consider the computational details of the technique, it will show how the model should be structured in order to use the technique, what the computational advantages of the technique are, and briefly discuss how the technique “works”. The reader is referred to Bloom et al. (1984), Desrochers et al. (1986), or Pereira and Pinto (1985) for examples of both actual applications and mathematical details of the Bender algorithm.

The structure of a model to be solved using Bender decomposition should be block-diagonal, with the overwhelming majority of the off-diagonal coefficients in the constraint matrix being equal to zero. The reason for this is that the technique consists of solving a series of small LP's almost as though they were independent of one another, then iteratively re-solving them using the previous iteration's solutions to each small problem as a means for modifying the constraint set for the current iteration. The method works best when there are no columns in common between the subproblems and few constraint rows in common, with few constraints crossing more than two subproblems.

The structure of the full model and an annual subset are shown schematically in Figures 4.1 and 4.2, respectively. All the non-zero elements in the matrices are contained within the objective function "boxes" and the annual or monthly boxes. Although neither figure is drawn to scale, it can be seen that most of the off-diagonal elements in the large problem are zeroes, and that the annual and monthly models do not have any columns in common. They are therefore block-diagonal and conform to the other specifications already noted. If the figures were to scale, it would be possible to see that only a small proportion of the rows cross the annual and monthly boundaries, and that no rows cut across more than one time period. The actual constraints that do cross the boundaries are fish stocks "moving" from one time period to another, along with storage levels in the hydrosystem and installed capital equipment (with a lifetime greater than one year) for the various management alternatives. The decomposition can be further extended within months to divide the hydrosystem and from the fish behavior, since the only link between these is the flow patterns within the system.

The computational advantages are a result of two factors already noted – round-off error and CPU time. Round-off error is reduced by solving a series of relatively small LP matrices. CPU time is reduced for the same reason. To see why the second point is true, consider the following example. Suppose one has an LP with a constraint matrix of 10,000 rows, and that the CPU time

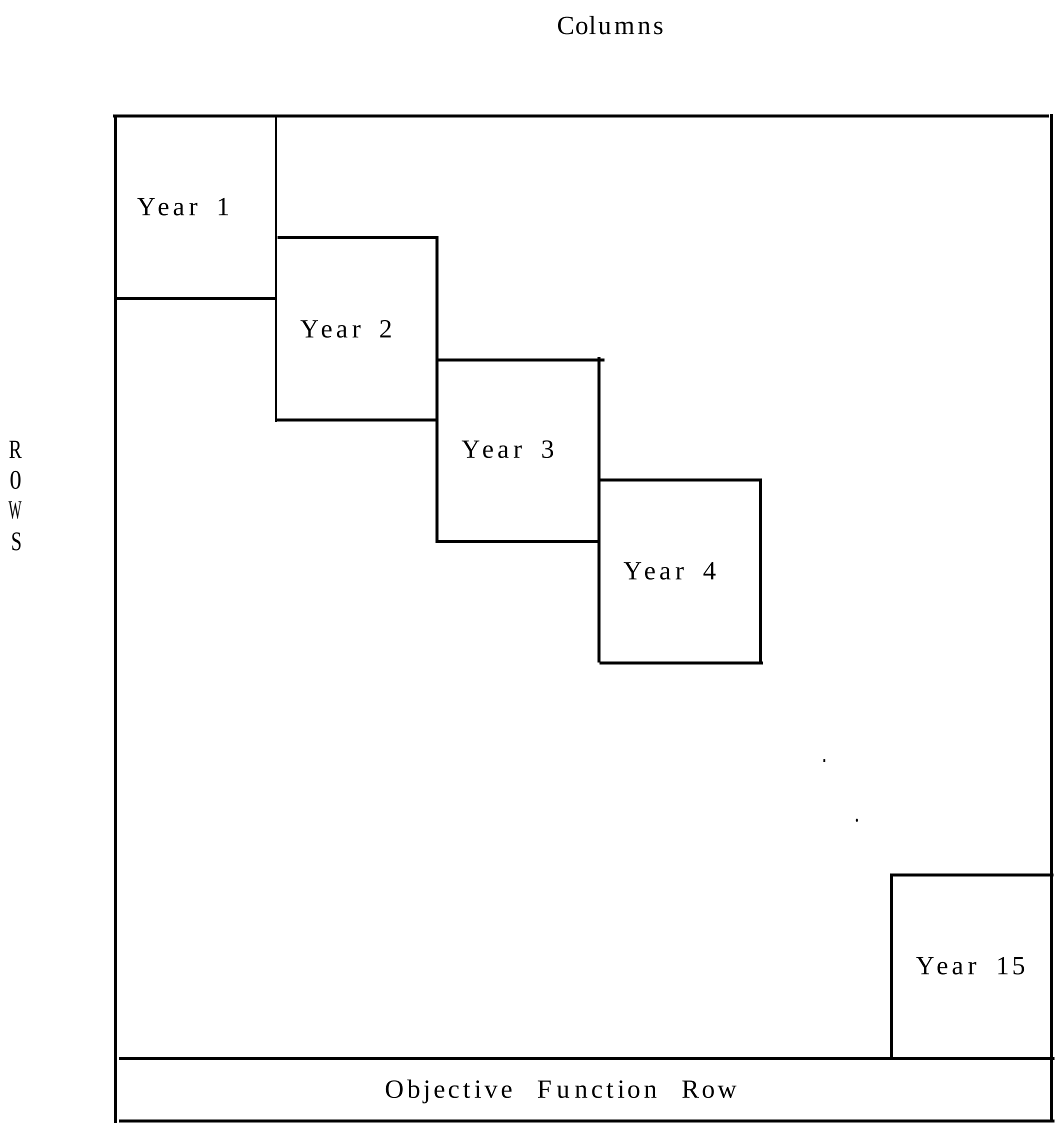

Figure 4.1. Full Model 15-Year Matrix

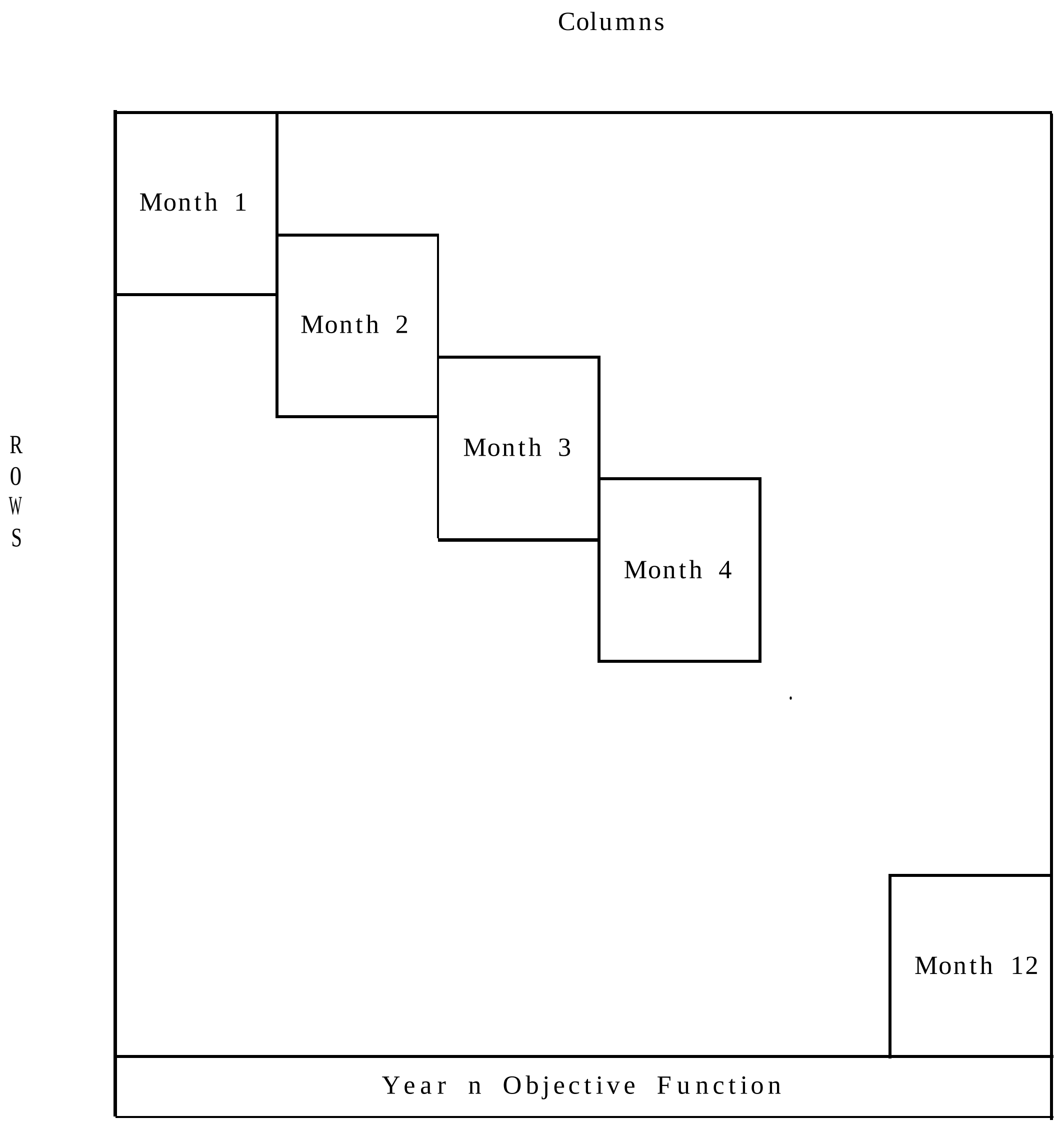

Figure 4.2 Full Model Annual Matrix
(Inputs from previous years not shown)

to solve it is equal to m^3 milliseconds (using the above rules of thumb), where $m=10{,}000$ in the example. Then the CPU time is equal to $(10{,}000)^3/1000$ seconds , or 10^9 seconds, or 30 years. Instead, consider the same constraint matrix decomposed into 100 subproblems of 100 rows each. Assume that the same m^3 rule applies to the subproblems, and that 10 iterations of the overall problem are required. Then the CPU time, in seconds, to solve the problem is $(100^3$ milliseconds per subproblem * 100 subproblems * 10 iterations), or 10^6 seconds (about 10 days), a time savings of three orders of magnitude. While the solution time involved is still very large, it is clearly much faster to solve the series of small models than to solve the single large model, under the assumption that the number of overall iterations is comparatively small. These are, of course, simply examples chosen to illustrate the principles involved, and should not be taken to be indicative of the time needed to solve the proposed Columbia Basin LP. We do expect that the decomposed models described below will be feasible to solve on mainframe computers or fast minicomputers, although the actual amount of time required remains to be determined.

There are disadvantages to using these decomposition techniques, of course. The primary analytical disadvantage is that one does not, strictly speaking, achieve an optimal solution with them. In practice, however, examples of applications of Bender decomposition converge rapidly, and it is easy to calculate upper and lower bounds for feasible solutions, so that one can determine how far the trial solution objective function value is from a truly optimal solution at any iteration. A practical problem is that there are no “commercial-quality” packages available for solving LP's using Bender techniques, but the theoretical foundations are well established, and there have been numerous applications using real-world data in the operations research literature (see the references noted above). A final potential problem is that there are no automated algorithms for partitioning the constraint matrix to make it amenable to the decomposition solution methods, but the algorithms are well established, and it will probably be possible to adapt existing programs to fit the Columbia model. On balance, it is our tentative conclusion that the Bender techniques may be a method for finding a solution to the optimization problems posed for this project.

Aside from the advantages already noted, there are some simplifications that can be applied to the model if it is partitioned correctly. In particular, it should be possible to eliminate the overwhelming majority of the integer variables in the model described in Chapter 3, by separating the smolt migration from the hydrosystem model. This can be accomplished because if the hydrosystem modules are separate from the smolt migration, then the flows from the hydrosystem can be passed as exogenous constraints to the smolt migration module, where most of the integer variables in the overall model are found. If, instead of being endogenous activities in the smolt migration, the flows are treated as exogenous constraints, then it is no longer necessary to use integer variables to accurately represent the effects of variable flows rates through reservoirs and variable flow proportions at dams on the downstream migrants, although the flows will need to be transformed into migration rates and proportions passing over spillways and through turbines. This greatly increases the probable feasibility of solving the overall model.

The detailed workings of the Bender decomposition method are beyond the scope of this chapter. However, in order to give the reader a general idea of what happens in solving a decomposed problem, we will attempt a broad, heuristic explanation of the method, using a simple example. The example follows a problem explained in detail by Pereira and Pinto (1985). Their article contains an appendix with a detailed exposition of the Bender decomposition as applied to a water resources problem.

The method can be explained intuitively as follows. Suppose one has an LP that can be broken into (say) two parts. The parts have a small proportion their of constraints (rows) in common, and no activities (columns) in common. This problem, therefore, has an objective function that can be divided into the costs for the first part of the problem, and the costs for the second. The constraints can be divided into those that are unique to each part of the problem, and those that are shared by each of the two parts. The Bender decomposition method is an algorithm that allows the iterative solution of the two sub-problems that assures convergence, within upper and lower bounds that are known after the first iteration, to the “true” optimal solution that would be obtained if one solved the entire problem as a single unit.

Suppose, for example, one has a power production optimization consisting of the operation of a mixed hydro and thermal plant system, where the first “stage” is operations in year one and the second “stage” is system operations in year two. The problem can be divided into two stages, based on operating year, with the objective being to minimize the sum of operating costs for the two years, subject to constraints on the minimum power produced in each of the two years. While each year has its own set of constraints, the two years have some constraints in common, namely the storage levels in reservoirs at the end of the first year. This example fits the above description of the decomposition method reasonably well.

The method for solving the problem works as follows. One first minimizes the cost of operation for the first stage (year), ignoring the second year entirely. This gives a lower bound of the objective function value for the overall problem, since by assumption second-stage costs are zero. It will also, in all likelihood, leave reservoir storage at the end of the first year at a much lower level than would be the case if the second year’s operations had been considered. Next, one solves the second stage problem, taking the beginning-of-year reservoir levels from the solution of the first-year problem as fixed constraints. In this example, the probable result is that the thermal plants will operate at a level that is "too high” (since reservoir levels at the beginning of the year were too low) producing a solution that is more expensive than would be optimal. This gives an upper bound on the possible objective function value for the overall problem. In Figure 4.3, the value of the overall objective function is given by the upper and lower points at iteration one.

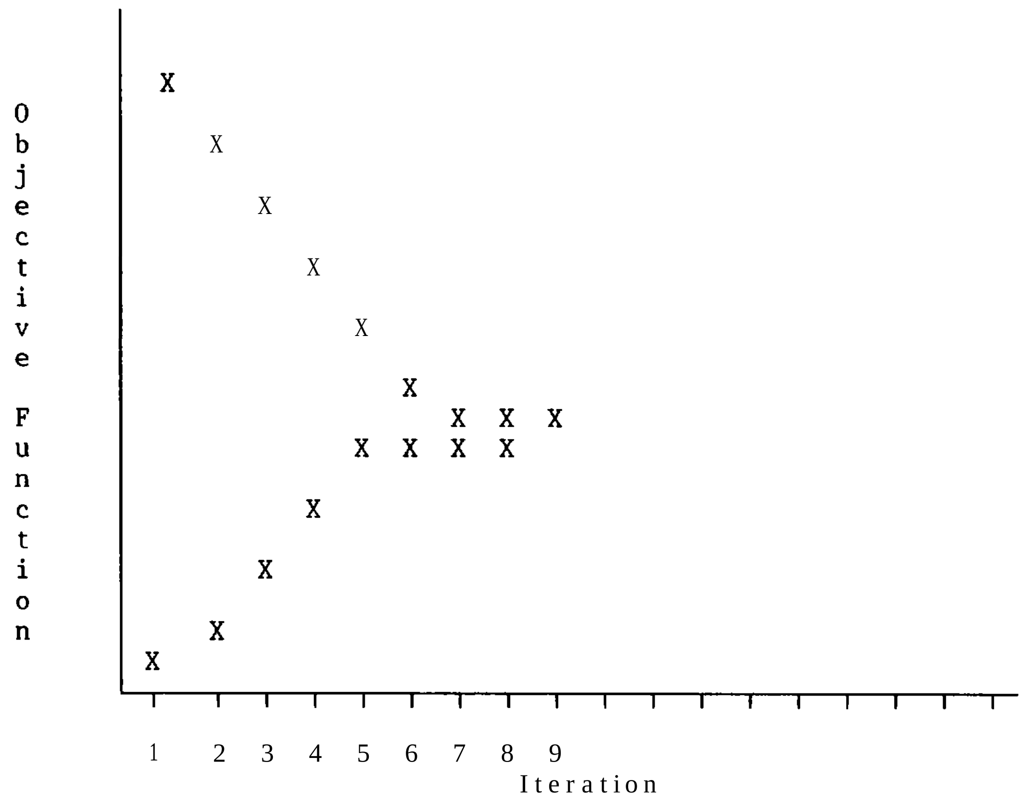

Figure 4.3. Convergence of Objective Function Values in Bender Decomposition

The next step in the process is to revise the first stage problem, based on the results in the second stage, and re-solve the first stage problem. This is done using the dual values from the second-stage solution to "tighten" the constraints on the first-stage problem. In the example this would probably mean requiring the first year model to leave more water in storage at the end of the (first) year. Then the second-stage problem is solved again, with constraints from the second solution of the first-stage problem that will be more nearly "correct" than the constraints used for the first solution of the second-stage problem. The process is repeated until the objective function converges, within exogenously specified limits, on an optimal solution to the overall problem.

Conclusion

The model outlined in the previous chapter will require a great deal of data, but that data is nearly the same as the information needed for the system simulation model. The very large size of the model makes it infeasible to solve it using commercially available mathematical programming packages. The model will need to be re-structured in order to make it possible to solve and produce useful answers for policy analysis. Based on a review of recent modelling efforts in both the hydrosystem and electrical power fields, it is our conclusion that it may be possible to solve a detailed, large-scale mathematical programming model for the Columbia River Basin. However, the very large size of the model makes the application of the techniques outlined above experimental in nature. We cannot guarantee that attempts to build and solve the model will be successful.

Chapter 5

Assessing the Opportunity Cost of Fish Mitigation Activities in the Columbia River Basin

INTRODUCTION

This chapter examines a variety of methods for assessing the opportunity costs of fish mitigation activities. It is specifically concerned with the opportunity costs associated with irrigation and hydropower.

In the case of irrigation, we assume that a water rights market exists in the region, such that farmers could buy and sell rights to irrigation water. Although such rights markets are in use elsewhere in the Western states, no markets for irrigation water are presently in operation in the Pacific Northwest. While we do not advocate the establishment of such markets, previous work in the area suggests that both farmers and electricity consumers might both be made better off if in fact such markets were to function. The fact that irrigation withdrawals have been established to be important factors limiting fish production in some subbasins in the region, including the Yakima and Umatilla, suggests that reduction of irrigation withdrawals might well be a cost-effective means for enhancing fish runs. The work in this chapter should be viewed as steps toward developing a means for establishing the difference in costs between enhancing fish runs with and without an active water rights market, in which rights might be acquired for enhancing fish runs. It should not be taken as advocacy of establishing such markets.

METHODOLOGIES FOR QUANTIFYING OPPORTUNITY COSTS

Opportunity Cost of Irrigation Water

Water withdrawn from the Columbia river system is one of several inputs into the agricultural production processes in the Columbia River Basin. Two sets of forces determine the amount of irrigation water used. The first includes physical factors such as the specific requirements of the crops and the drainage potential of the soil. The second set of factors is economic and include the cost of irrigation, the costs of other inputs used in the production process, and the prices received by farmers for their produce. The paper by Diewert (1985) on measuring the economic benefits of infrastructure services develops the theoretical basis for the models reviewed in this chapter.

To briefly summarize the microeconomic theory, the opportunity cost of the marginal unit of surface water to the agricultural sector can be determined from econometrically estimated restricted profit or cost function models. These estimates are, by construction, partial equilibrium or short run measures since the technology and level of other arguments in the model, such as irrigation water, are being held fixed. The restricted profit function assumes that prices of the variable inputs and outputs are constant. This constant-price estimate of the opportunity cost of water is a reasonable approximation of the gross effects if changes in the quantities bought and sold by this sector are not large enough to influence market prices. In the restricted cost function model, the level of output, the prices of the variable factors, and the quantities of the fixed factors are held constant. The resulting constant-output estimates of the opportunity cost of water would be similar to the estimates of replacement costs if the replacement was to be done by existing firms assuming that output levels, input prices, and the level of irrigation water are held fixed. It can also be shown that under certain conditions the firm's willingness to pay functions for the set of quasifixed inputs (i.e. water) can be interpreted as a system of inverse demand functions.

Opportunity Cost of Water in Hydropower

In developing a framework to quantify the opportunity costs of water diverted from hydropower generation to aid fish passage, we are left with a conceptually difficult job due to the structure of this sector, the publicly regulated pricing structure, and the current surplus power that exists in the region. Ideally we would like to assess the increase in the marginal cost of producing a given amount of energy, or to value the lost output at its social cost. However, the net benefit or cost measures based on the restricted profit and restricted cost functions assume that the sector is competitive, and that a well-defined, twice continuously differentiable cost function exists. These conditions do not describe the hydropower sector.

The conventional approach to opportunity cost measurement has been to determine replacement cost for the energy foregone as the result of a change in the water allocated to hydropower activities. The marginal cost of generating energy beyond some “base case” level becomes significantly greater as the relatively cheap hydropower energy is replaced by the more expensive sources, which include thermal or coal-powered plants. Eckstein (1961) suggests that we should use "the cost of providing comparable output by the cheapest alternative means . . . however alternative cost computations are not valid substitutes for estimated opportunity cost unless there is a clearly defined objective which is going to be met in one way or another” (pp. 52-53). For example, if power capacity already exceeds the demand at the going rate structure, the marginal value of an additional unit of power is going to be less than the cost of alternative capacity, since the alternative would not be constructed; the extra power could only be marketed at a lower price-- possibly the price at which it is sold to the Southwest. Thus the use of replacement cost to measure opportunity cost really is equivalent to the problem of choosing an alternative means of minimizing the cost of achieving a predetermined objective, i.e., producing an equivalent amount of power.

The difficulties in quantifying the replacement cost include determining what type of power is being sacrificed and deciding how to value the lost energy. To determine the quantity of hydropower that is lost requires knowledge of the relationship between stream flow and generating capacity. Variations in the annual volume of streamflows alter the regional cost of electricity production because of a change in the mix of cheap hydropower and other more expensive sources of energy. The Nor thwest Power Planning Council

(NPPC) reports that in the early 1980's hydropower operating costs were approximately 10 percent of new thermal power costs.

Primary or firm energy is lost if the depletions occur during critical water conditions. Secondary energy is lost if the depletions occur only when stream flows exceed the critical water conditions. Peaking capacity may also be affected if the diversions reduce the maximum amount of power that can be produced under an adverse combination of loads and streamflows.

Fish flow enhancement due to the Water Budget represents an additional demand for instream flows. Allocation of water to this activity may result in curtailment of water available for hydropower and/or irrigation in all years, including the critical flow year. Since the fish enhancement flows for the Water Budget are needed every year, this commitment reduces the critical flow; power planners have less flow to count on in the event that a critical flow year is realized. It has been argued, therefore that the lost power should be valued at the replacement cost for firm power. However, if there is a surplus of firm power over and above demand at prevailing rates, economic theory would indicate that the output lost due to Water Budget flows should be priced at its value to society, which in high flow years might be the rate at which the power is sold outside the region, or may even be zero if there is no market for the power outside the region.

All of this suggests that the value of the lost power cannot be tied to a single rate, but rather should vary over time because of the possibility of substituting surplus firm or nonfirm hydropower for thermal power. These substitutions would result in savings in variable costs (due to thermal generation avoided), which can be translated into lower replacement costs and lower opportunity costs. In most of the empirical work to date (see next section) the opportunity cost of the lost hydropower is valued at 35 m per KWH, which is the reported replacement cost for a thermal plant. While this may be defensible for firm power replacement when the hydro system is operating at full capacity it appears to be less justified during an period of excess capacity. The present glut of both firm and nonfirm power suggests valuing the lost energy at the rate at which surplus power is being sold outside the region.

REVIEW OF SELECTED EMPIRICAL ANALYSES

A substantial amount of research has been undertaken to evaluate the irrigation-hydropower tradeoff in the Pacific Northwest. The large proportion of the economic efforts have produced estimates of the cost of future irrigation development in the region in terms of lost hydropower. The context of the current project does not concern the benefits and the costs of future irrigation development *per se*; rather the problem is how to cut back or reallocate existing market uses of the stream flows. The problems are not symmetric, although information contained in these studies may be useful in characterizing the current status of the hydropower and agricultural sectors.

The objective of the next three sections is to provide a brief review and comparison of the empirical evidence regarding the economic value of water used in irrigated agriculture in the Pacific Northwest, with an emphasis on the Yakima and Umatilla areas. The estimates are obtained from a careful review of existing research in the area; no new estimates or analyses have been under taken. The next-to-last section of this chapter uses these estimates to project prices and total costs for purchase of irrigation rights in the Umatilla and Yakima subbasins, under the assumption that water rights markets were instituted in the region.

The studies are organized by the methodology used to obtain the empirical results: econometric or programming techniques. The different methods and data sets used in calculating values results in a set of values which are not directly comparable. Some values pertain to a short run time frame, while others reflect longer run values. In addition, some studies report a marginal value for water, while other estimates may reflect an average value of water; values may also be crop-specific or relate to a mixture of crops. Finally, the values may be either on-site (at the point of application) or instream water values. Where possible, these differences are carefully delineated.

Econometric Studies of the Economic Value of Water

Econometric analyses of agricultural production functions or the dual cost, profit, or revenue function models can be used to calculate a marginal value for the water input. If all other units are held constant, the marginal physical productivity of water (i.e., change in crop output as a function of change in irrigation water applied) used on the specific crop or mix of crops can be calculated from information on the production function. The marginal value of water is simply the marginal physical product times the appropriate market output price. Since these estimates are primarily made using farm-level or county-level data, the marginal value products reflect "on-site" values for water. Furthermore, the econometric analyses can produce either short-run or long-run estimates depending upon the specification of the production technology.

Econometric techniques can also be used to estimate the shadow value of irrigation water using a short run or temporary equilibrium model. The estimates of the value of water that result from using a restricted cost or restricted profit function model (where the quantity of irrigation water is held constant) are, by construction, short run measures since the technology and level of other inputs and crop output are being held fixed. In the econometric studies reviewed, the econometric approaches tend to be rather dated, using information that in some cases is more than two decades old. They are examined below in order to give the reader an idea of how these methods could be usefully applied to more up-to-date information, and to compare the strengths and weaknesses of the econometric and programming approaches.

Holloway and Stevens (1973)

The oldest of the econometric studies is one by Holloway and Stevens. It is based on an aggregate production function analysis of water resource productivity in the Pacific Northwest using cross-sectional data at the county level for 1964. The purpose of the study was to evaluate the economic efficiency of the water allocation that existed in 1964 and to make inferences about potential irrigation development in the region. While it is based on very dated information, the general approach employed in the study may be useful in producing more timely estimates of prices in a potential regional water rights market. Estimates of irrigation productivity were obtained for selected areas within the region, each area consisting of counties that by assumption are homogenous production units .

The Washington-Oregon-Idaho study area was divided into five county groups, designed as follows:

Area A: 41 counties, which typically produced field crops and livestock products;
Area B: 15 counties, which produced primarily livestock and livestock products;
Area C: 20 counties, which produced mostly field crops;
Area D: 27 counties, which produced mostly livestock and dairy and livestock products: and
Area E: 16 counties, which had highly diversified production.

Both the Yakima basin in Washington and the Umatilla basin in Oregon are part of the Area A county group. More specifically, the counties in this group had greater than 50 percent of the total value of farm product sold from field crops and livestock and livestock products, and no less than 20 percent from each product group.

The production function for each region includes eight variable input aggregates - labor expenses, operating expenses, equipment and buildings, cropland, animal capital, irrigation water, farm investment in drainage, and farm investment in water conservation. The output variable was the total value of farm products sold. Data were primarily taken from the 1964 Census of Agriculture; irrigation water data were estimated using 1959 Census of Irrigation application rates and the number of irrigated acres reported by the 1964 Census of Agriculture. Information on drainage investment and water conservation practices were constructed by Holloway and Stevens based on historical records of farmers participation in Agricultural Conservation Program cost sharing arrangements, as reported in the annual statistical reports of the Agricultural Stabilization and Conservation Service. These two variables – the service flows from drainage investment and water conservation practices – were included with the service flow for equipment and buildings in the final regression analyses.

Both linear and exponential functional forms for the aggregate production function were estimated. The exponential form is theoretically more plausible since it allows for diminishing marginal productivity. Holloway and St evens, however, utilized the linear functional form stating that "(1) the statistical tests were more precise in the linear form; (2) the linear forms were good approximations of non-linear functions over the range of the data; and (3) MVP estimates from the linear form (for the irrigation variable) were not significantly different from the Cobb-Douglas function." (p. 28) The Holloway and Stevens results are presented in Table 5.1.

They found that significant differences existed within the five areas with respect to the estimated MVP per acre foot of irrigation water. The returns were significantly higher for field crops (area C) than in the livestock and dairy product areas (areas B and D). Furthermore, the areas in which water was the most productive in 1964 had the greatest potential for future irrigation development, southeast Washington and southeast Idaho.

Since the Yakima and Umatilla areas fall within Area A, their estimates of the MVP of irrigation water provide the best measures of the MVP for these two areas. The values reported in Table 5.1 reflect the values at the delivery point in 1964 dollars. Holloway and Stevens estimate that at the application point the MVP for Area A using the linear model increased to \$10.150 per acre foot, assuming 75% irrigation efficiency, and to \$15.226 per

Table 5.1. Holloway and Stevens (1973) Estimates of Marginal Value Product (MVP) of Water, Based on Linear Regression[a]

($/acre foot of water delivered, 1964 $)

Area[b]	Estimate of Marginal Value Product of Water	90 percent Confidence Interval
A	7.613	3.510 - 11.716
B	4.928	3.681 - 6.177
C	10.590	7.316 - 13.864
D	3.403	0.069 - 6.737
E	10.688	-3.754 - 25.130

Notes

a. Fitted regression: $Y=f(X_1,X_2,X_3,X_4,X_5,X_6)$ where Y=value of farm products sold and inputs 1 through 6 represent labor, operating expenses, service flow on capital, cropland, animal unit months of available grazing, and irrigation water, respectively.

b. Definition of area: A--41 counties in central and NE Oregon and southern Idaho (field crops and livestock products); B--15 counties in eastern Oregon (primarily livestock products); C--20 counties in SE Washington and SE Idaho (field crops); D--27 counties in coastal Washington and Oregon (livestock and dairy products); and E--16 counties in Willamette-Puget Trough (diversified).

Source : Holloway, M., and J. Stevens. 1973. "An Analysis of Water Resource Productivity and Efficiency of Use in Pacific Northwest Agriculture", Special Report 383, Agricultural Experiment Station, Oregon State University.

acre foot assuming 50% irrigation efficiency. These latter estimates were generated by the linear regression equation when the quantity of the irrigation variable was divided by .75 and .50, respectively. Although the data used in the study are too old to be enable one to reach any conclusions about prices in potential water rights markets, it is illustrative of the methods and results that one might expect from a more current study.

Frank and Beattie (1979)

This work, along with the study by Gibbons, below, is included in order to show how prices for irrigation water in the Columbia might compare with those in other regions in the West. Since there are active water rights markets elsewhere the results reported below may give some general indications as to how prices in the Columbia Basin would look relative to other regions. As with the Holloway and Stevens study, the results shown below should be taken as illustrative examples.

In a study entitled "The Economic Value of Irrigation Water in the Western United States: An Application of Ridge Regression", Frank and Beattie provide a means for comparing regional water values. Their research is a comprehensive study based on a single estimation technique and data source, designed to discern regional economic characteristics and differences of irrigation water demand in the West.

Eleven homogenous regions, including the Columbia-Snake River Basin, were the major irrigated areas. Agricultural output, in dollar-value terms, was specified to be a function of nine variable inputs: irrigation water applied, value of land and buildings, hired labor expenditures, fuel expenditures, fertilizer and lime expenditures, feed expenditures, value of machinery inventory, value of livestock inventory, and miscellaneous expenditures. The data were obtained from the 1969 Census of Agriculture.

Each regional Cobb-Douglas production function was fitted using ordinary least square (OLS) and ridge regression techniques. Frank and Beattie report that the OLS estimates were highly unstable due to a high degree of multicollinearity, and had theoretically incorrect signs. The ridge regression estimates were more compatible with theoretical expectations, and had lower standard errors.

From the fitted production functions, the demand for irrigation water was derived for the long run, for two intermediate runs, and for the short run. Marginal irrigation water values were estimated for 1969 at the mean values of water usage, other input levels and variable input price. These water values are reported in Table 5.2. The marginal value of water in irrigated agriculture is lowest in the Snake-Columbia Basin in the short run.

For all lengths of run considered, Frank and Beattie assumed that the livestock related inputs were constant, since it was believed that changes in livestock activity would cause virtually no change in increased irrigation of pastureland. The long run refers to the length of run when all crop related inputs are treated as variable; the intermediate run I and intermediate run II characterize production when only land, and then when land and machinery, respectively, are held fixed. In the short run, all crop-related inputs except water are considered fixed to the firm.

With respect to the Frank and Beattie study, it should be noted that the marginal water values are reported at the mean of all explanatory variables and that the point estimates depend upon the level of the input levels, input prices, and product prices. Frank and Beattie also state some concern with the fact that the longer run water values generally exceeded the shorter run values. They suggest that this may be due to the fact that either too much water (on average) was applied in 1969 or that the mean levels of short run fixed factors were not optimal. Note that if all factors including water were fixed at <u>optimal</u>, profit-maximizing levels, the marginal values would be the same for all lengths of run.

It would have been helpful if Frank and Beattie had reported the standard errors associated with their estimated water values. This would

Table 5.2. Marginal Water Values Under Alternative Lengths of Run, 1969$[a]

Region	Length of Run			
	Long Run	Intermediate Run I	Intermediate Run II	Short Run
Snake-Columbia Basin	1.80	1.80	1.83	1.71
Central California	27.79	6.79	5.30	4.92
Desert Southwest	b	22.74	10.37	7.73
Upper Colorado Basin	8.08	6.87	6.65	7.02
Upper Rio Grande Basin	---	22.07	2.52	2.25
Lower Rio Grande Basin	10.00	8.94	6.54	3.51
Upper Missouri Basin	5.98	4.40	4.27	4.09
Northwestern Ogallala	b	24.88	11.97	9.42
Northwestern Ogallala	16.58	12.71	8.40	6.63
Central Ogallala	27.74	21.53	7.44	4.36
Southern Ogallala	8.81	8.27	3.08	2.03

Notes

a. Based on mean levels of all variables for each region.

b. Values not reported--see text.

Source : Frank and Beattie (1979), Table 6.

have enabled some cross study comparisons of confidence intervals around the point estimates, rather than simply comparing point estimates.

Gibbons (1986)

In another inter-regional comparison, Gibbons reports on a series of studies funded by the US Department of Agriculture to estimate crop water production functions in six states: Arizona, New Mexico, Texas, California, Washington, and Idaho. The production functions were estimated under controlled conditions. The marginal values are shown in Table 5.3 for a percent reduction in the amount of water applied from the yield maximizing levels for that particular crop experiment. The prices used to value the marginal productivities were the 1980 national averages by crop.

The estimates reported by Gibbons seem rather high relative to those reported by Frank and Beattie. This may be partially due to: (1) the experimental nature of the production function estimates and the fact that the values are reported at the yield maximizing levels; (2) the data reflect 1980 prices as opposed to 1969, 1974 or 1978 prices; and (3) the data are crop specific. As with the Frank and Beattie study, the Gibbons work should be viewed as an example of how one might carry out inter-regional comparisons, perhaps a a “check” on the results of future studies in the Columbia Basin proper.

Programming Studies of the Economic Value of Water

Probably the most extensively used technique for discerning water values is linear programming. In this approach, a profit function is maximized (or cost function is minimized) subject to various economic and physical constraints. A solution of this type yields estimates of the marginal value productivities (shadow prices) of each constraining input in addition to the optimal input combination. One can relax the water supply constraint(s) and determine the set of shadow prices for additional units of water.

Table 5.3. Marginal Water Values from Crop-Water Production Functions, 1980 $ per acre-foot

Crop	Value					
	Idaho	Washington	California	Arizona	New Mexico	Texas
Grain sorghum				<15		113
Wheat		59		22		35
Alfalfa				25	25	
Cot ton			71-129	56	61	
Corn					52	57
Sugar beets	144					
Potatoes	698	282				
Tomatoes			390			

Note: Values have been calculated at 10 percent reductions from yield-maximizing water levels. In each case, 1980 prices, average efficiency, and medium-textured-soil functions have been used.

Source : Gibbons (1986), Table 2-2.

Average water values can also be estimated by deriving a series of solutions for a range of water costs, holding all other constraints constant. The solutions yield the profit maximizing combinations of inputs and outputs, including total irrigation water, for each water cost. The set of solutions maps out a demand schedule for irrigation water.

More recently, programming models have been designed which allow for both alternative levels of water usage and of irrigation technologies.

Houston (1984) and Houston and Whittlesey (1986)

In the Houston (1984) dissertation, a two-stage programming model is developed to evaluate irrigator production and regional price responses to water and energy conservation policies. The first stage focuses on the decision made at the producer level regarding levels of input usage and output mixes. Irrigators choose cropping and irrigation mixes and rates at expected commodity prices under regional resource constraints.

The second stage utilizes the producer-level solutions in a regional allocation model to determine regional level effects of changes in price and rationing policies for region resources, including inter-area transfers of conserved water and energy. Baseline irrigated acreage, water, electricity, production and crop prices are estimated from 1982 state crop production data.

The programming model allowed for changes in water policies; the opportunity value of water for irrigation was compared to its instream hydrogeneration value at each diversion location. In the water pricing scenario, irrigators were assumed to have fully transfer-able water rights and would sell water which had a higher- marginal value for- hydrogeneration than for agricultural production. The ins trean hydrogenerat ion value was allowed to vary from zero mills per KWH to 50m per KWH. These results are shown in Tables 5.4 and 5.5.

Table 5.4. Hydropower Production from Water Sold and Irrigation Electricity Use in GWH at Alternative Hydropower Values for Selected Production Areas

Surface Water Production Area	Hydropower from Conserved Water		Baseline Irrigation Electricity use	Percent Irrigation Electricity Conserved	
	20m/KWH	40m/KWH		20m/KWH	40m/KWH
Ferry-Stevens,WA	0	13.51	19.85	12.49	29.37
Columbia Basin,WA	0	603.55	419.46	3.48	3.75
Wenatchee, WA	0	.28	69.60	.45	.47
Big Bend East,WA	.29	.29	59.69	0	0
Deschutes, OR	0	.50	151.98	0	0
John Day, OR	.03	.03	15.93	0	0
Weiser, ID	34.12	48.68	14.92	21.65	51.07
Boise, ID	87.85	212.20	57.82	8.09	72.28
West Side, ID	138.43	272.92	88.04	9.64	49.14
Neely-Milner, ID	138.67	150.01	30.44	52.23	53.61
Regional Total	592.00	1,518.00	5,660.00	1.17	3.09

Source: Houston, J.E., and N.K. Whittlesey. 1986. "Modeling Agricultural Water Markets for Hydropower Production in the Pacific Northwest", Western Journal of Agricultural Economics, 11(2): 221-231.

Table 5.5. Regional Response to Increasing Hydropower Values of Conserved Water
(1985 Electricity Rates)

Type of Response	Million Units	Hydropower Value (m/KWH) 0	10	20	30	40
Net returns to irrigation[a]	$	1,397	1,396	1,396	1,398	1,408
Value of water sales	$	0	5.50	19.60	48.18	76.24
Net farm income[b]	$	1,397	1,402	1,416	1,446	1,484
Consumer surplus	$	2,692	2,692	2,686	2,670	2,648
Total irrigated acreage	acres	8.076	8.076	8.059	7.987	7.902
Water diversion	ac.-in.	239.66	236.97	230.06	221.42	216.64
Hydroelectric power from water sales	KWH	0	162	592	1,218	1,518
Irrigation electricity use	KWH	5,660	5,638	5,594	5,565	5,485

Notes:

a. Net returns to irrigation includes only the value of sales from agricultural commodities.

b. Net farm income includes the value from crop sales plus the value from water- sales.

Source: Houston, J.E., and N.K. Whittlesey. 1986. "Modeling Agricultural Water Markets for Hydropower Production in the Pacific Northwest", Western Journal of Agricultural Economics, 11(2): 221-231.

In the water rationing scenario, shown in Table 5.6, the availability of Columbia River instream flows was parametrically decreased by 5 percent decrements. Irrigators received no compensation from reduced water rights, but the model yielded “potential” payments through shadow prices of the agricultural water use or the change in net returns to producers using irrigation. The results of the water rationing scenario are used in the illustrative application to the Yakima and Umatilla subbasins, below.

Houston’s regional solution results for both water policy scenarios should be interpreted as short run reactions or responses, since commodity prices to farmers are held constant at pre-season 1982 expected levels. Under the water pricing policy, the net returns to irrigated land increased by $12 million and the net benefits to farmers by $43 million when water was exchanged at 40 mills per KWH. The reduction in irrigation water at this opportunity cost was about 10% of baseline Columbia River diversions. Houston states that these results were not uniformly distributed across all producers in the Columbia River Basin. Irrigators in downstream areas of the Columbia River Basin were relatively unaffected by changing hydropower values, as the potential generation of electricity from conserved water in these areas was relatively minimal.

Under the water rationing scenario, regional net farm incomes rose despite the lack of compensation for water right losses. The higher returns resulted from higher equilibrium product prices which resulted from the solution to the second stage of the programming model. For example, potato prices rose nearly 38 percent with only a 10 percent reduction in irrigation water while apple prices rose by 15 percent when water availability was decreased by 15 percent. The location adjustments also differed from those associated with the water pricing scenario, and more importantly, inefficiencies resulted from the failure to incorporate locational differences in the value of a unit of water to hydropower generation. Production areas in the lower river basin received the same cutbacks as areas upstream which had higher power- tradeoffs. The shadow values reported in Table 5.6 illustrate these discrepancies. In addition, the concentrated

Table 5.6. Shadow Prices of Columbia-Snake Water in Selected Production Areas Under Regional Columbia River Water Rationing

Production Area	Columbia River Surface Water Allocation				
		Percentage Reduction from Full			
	Full	5	10	15	20
	------------- ($ per acre-inch ------------				
Methow, Okanogan	1.35	1.35	1.35	11.98	11.98
Wenatchee, Chelan	0.82	9.66	9.66	18.58	18.58
Yakima	0.00	4.27	4.27	5.70	5.70
Northside Columbia	0.00	0.00	2.03	10.08	10.08
Ferry, Stevens	0.82	0.82	0.82	0.82	2.30
Columbia Basin	1.61	1.61	1.87	1.87	1.87
Big Band East	4.56	4.56	4.56	4.56	4.56
Pend Oreille	0.00	1.27	3.71	3.71	3.71
Spokane, Kootenai	0.00	0.00	0.00	0.00	0.00
Palouse, Lower Snake	0.83	0.83	0.83	11.35	11.35
Walla Walla	2.02	2.02	6.53	7.69	7.69
Hood	14.61	14.61	14.61	24.04	24.04
Deschutes	4.88	5.32	5.36	5.88	5.88
John Day	0.00	4.89	4.92	5.38	5.38
Umatilla	2.24	7.15	7.35	10.35	10.35
Grand Ronde	10.42	10.56	10.57	10.73	10.73
Burnt, Powder	6.95	7.51	7.55	8.21	8.21
Owyhee	1.08	1.08	2.78	2.78	2.78
Clearwater, Salmon	0.00	1.96	3.68	3.68	3.68
Upper Salmon	1.50	1.50	1.50	1.50	1.50
Weiser	0.00	0.58	0.58	0.58	0.58
Payette	0.36	0.36	0.36	1.79	1.79
Boise	0.75	0.75	2.24	2.24	2.94
Bruneau	0.49	0.49	2.72	2.72	2.72
West Side	0.63	0.63	2.26	2.26	4.39
Northern Streams	0.37	0.37	0.37	2.68	2.68
Neeley, Milner	1.22	1.22	1.22	1.22	1.22
Heise, Neely	0.00	2.90	2.96	4.23	4.23
Henrys, South Fork	2.04	3.69	3.86	6.40	6.40

Source: Houston (1984), Table 4.38

fruit growing areas such as the Yakima, Northside Columbia, Hood, Methow-Okanogan, and Wenatchee-Chelan, suffered substantially greater impacts of an across-the-board water reduction policy. These areas also had relatively little impact on hydropower potential that could have been gained, having lower cumulative hydropower potential at their points of diversion. These shadow values are useful as measures of the opportunity cost of water in the Yakima and Umatilla areas.

Whittlesey (1980) and Butcher et al (1986)

In a study showing the increases in the opportunity costs of irrigation withdrawals, Whittlesey, and Butcher et al., analyzed the tradeoff between irrigation and hydropower uses of the Columbia River system in the period from 1960 through 1980. They approximated the marginal value of the water by estimating the alternative cost of providing the energy lost due to diversions as well as the energy consumed because of irrigation pumping and related electricity use. Prior to 1960, they report that the opportunity cost of stream flow diversions, which is defined to be equal to the value of energy that could have been produced had the water been left to flow downstream through the hydropower generators, was small. This was due primarily to the lack of reservoir and turbine capacity; the surplus water was simply spilled over the dam.

From 1960 to 1980 the opportunity cost of an acre foot of water diverted to offstream uses increased as more dams were constructed, reservoir capacity increased, and the value of alternative sources of electricity increased from 2.5 m per KWH to 35 m per KWH.

McCarl and Ross (1985)

A programming analysis by McCarl and Ross was done to determine the costs borne by electricity consumers under expanded irrigation from the Columbia River. This study differs from the Whittlesey and Butcher- et al. study in five aspects: it takes into account the stochastic nature of yearly stream flows; it allows electricity consumers to alter their demand behavior in response to price changes; power lost per foot of cumulative dam head is also allowed to fluctuate in response to stream flow levels; methods of irrigation repayment for power used in diverting water are not fixed (varies according to model specification); and irrigation diversions are not held constant, but are allowed to be interrupted in low flow years. While the data used in the study are too old to rely heavily on the specific results, the methodology is perhaps closest that which might be employed in future research, if programming methods are chosen to investigate irrigation opportunity costs.

In general, the McCarl and Ross study verifies the results of Whittlesey and Butcher et al. and shows that electricity consumers and those paying for the pumping costs incur substantial welfare losses due to proposed irrigation projects. Even when producers pay all of the pumping costs, the loss due to hydropower diverted for the potential East High project is nearly $49 per acre foot, or $126 per acre developed. If the pumping costs are not absorbed by the producers, the losses are nearly double in this particular project (Table 5.7, Pricing model 2). For all four areas considered in the analysis, the annual loss of electricity is close to $100 per acre, and exceeds $200 per acre if the government delivers the water to the farm gate.

Unfortunately, McCarl and Ross did not provide any estimates of the economic value of water (in $ per acre foot) that would be useful to the immediate objectives of this chapter.

CROSS STUDY COMPARISONS

To provide for some limited cross-study comparisons, the water values reported for the most of the econometric and programming studies have been converted to 1984 and 1985 units. These are shown in Table 5.8. In an effort to maintain some consistency with the calculations done by Gibbons (1986), the adjustments have been made using the indices of prices received by farmers repot-ted in Agricultulal Statistics, 1979 and 1986. These

Table 5.7. Comparison of the Marginal Value Products with Estimates of the Private and Social Marginal Costs of Irrigation, Five Homogeneous Farming Areas, Pacific Northwest, 1964

	Area A	Area B	Area C	Area D	Area E
Private (and Social) Marginal Value Product					
At delivery point ($/Acre-Foot)	7.613	4.928	10.590	3.403	10.688
At application point assuming 75 percent irrigation efficiency[a] ($/Acre-Foot)	10.150	6.571	14.120	4.537	14.250
At application point assuming 50 percent irrigation efficiency[a] ($/Acre-Foot)	15.226	9.856	21.180	6.806	21.376
Private Marginal Cost					
At delivery point ($/Acre-Foot)[b]	5.07	1.87	2.32	6.26	6.69
At delivery point ($/Acre-Foot)[b]	1.44	.60	.89	1.68	2.14
At application point ($/Acre-Foot)[c]	N/A	N/A	2[illegible].00	N/A	23.00
Lower Bound of Social Marginal Cost					
At Delivery Point[d]	13.72	11.03	19.78	14.78	19.69

Source: McCarl, B.A., and M. Ross. 1985. "The Cost Borne by Electricity Consumers under Expanded Irrigation from the Columbia River", Water Resources Research, 21: 1319-1328.

Notes to Table 5.7:

a. MVP estimates generated by the linear regression equations when the quantity of the irrigation variables (X6) was divided by the constants .75 and .50, respectively.

b. Weighted average assessment per acre (including operation and maintenance costs) by area for 73 irrigation districts in Oregon in 1966. Weights were irrigable acres in each district. Conversions to a per acre-foot basis were made on the basis of average delivery rates reported by Oregon irrigation organizations in the 1959 Irrigation Census. These rates were 3.52, 3.15, 2.60, 3.72, and 3.13 acre-feet per acre for Areas A through E, respectively. Additional data for three individual Bureau of Reclamation projects in Oregon (Area A) indicated recent prices of $1.50 to $2.00 per acre-foot of water delivered. Prices for the water delivered to farmers in the Columbia Basin project (mostly in Areas A and C) range from $2.50 to $3.00 per acre-foot in recent years. Water prices for an estimated 25 percent of the irrigated acreage in the Snake River Valley of Idaho are usually $2.00 to $2.50 per acre-foot.

c. Available data for sprinkler irrigation costs per acre-foot in the Willamette Valley in Oregon (primarily Area E) and in the deep well irrigation of eastern Washington (primarily Area C) include pumping and distribution costs. These estimates are not representative of average farmer irrigation costs at the application point, but rather the upper point on a range of costs since other distribution systems are less costly. The average price for water at the application point would likely be considerably below these estimates.

d. Estimates based on data from Bureau of Reclamation projects in the region. These estimates include the original investment cost per acre, adjusted to 1964 prices, a one-hundred year life expectancy, and a 5 percent opportunity cost for underpredicted investment.

Source: McCarl, B.A., and M. Ross. 1985. "The Cost Borne by Electricity Consumers under Expanded Irrigation from the Columbia River", Water Resources Research, 21: 1319-1328.

Table 5.8. Cross-Study Comparison of Water Value, Converted to 1984 and 1985 Values[a]

Source		Water Value		
		Original Year	1984	1985
		------------------------$ per acre foot----		----------
A. Econometric Studies				
Holloway and Stevens (1973)	(1964)	7.61[b]	20.88	18.82
	(1964)	10.15[c]	27.40	25.38
Gibbons (1986)	(1980, wheat)	59.00[d]	51.33	47.79
	(1980, sugar beets)	144.00[d]	171.36	162.72
	(1980, potatoes)	282.00[d]	335.58	318.66
Frank and Beattie (1979)	(1978, short run)	3.33	4.39	3.80
B. Programming Studies				
Gibbons (1986)	(1980, hops)	10.00	8.70	8.10
	(1980, alfalfa)	10.00	11.00	9.20
	(1980, corn)	31.00	34.1	28.52
	(1980, wheat)	52.00	45.24	42.12
	(1980, pears)	78.00	134.16	119.34
	(1980, apples)	86.00	147.92	131.58
Houston (1984)	(1985, Yakima)[e]	--	--	51.24
	(1985, Umatilla)[e]	--	--	85.80

Notes:

a. Values for the original years have been converted to 1984 and 1985 using the prices received by farmers as reported in Agricultural Statistics. The prices used for 198.5 are preliminary. No adjustments have been made for changes in technology or changes in factor prices.

b. At delivery point.

c. At application point.

d. 10 percent reduction from yield maximizing levels.

e. 10 percent reduction from full water allocation under the water rationing policy.

adjusted values should be regarded as crude approximations at best, since many factors have not been taken into account. These include changes in technology, changes in product mixes, and/or changes in factor prices that have occurred between 198411985 and the time the original studies were undertaken.

Among the three econometric studies, the Holloway and Stevens study most closely reflects economic (marginal) values for the subbasins of interest. However, the main limitation of the adjusted estimates from the Holloway and Stevens study is the fact that adjustments have not been made to a 20 year old data set for technology, input price and output mix changes. The results reported by Gibbons are based on controlled experimental data and represent a 10 percent reduction from yield maximizing output levels. This bears no relation to an economic value.

Of the two programming studies, the Houston study is perhaps the most up-to-date and relevant for the objective at hand, although the McCarl and Ross study is perhaps preferable on methodological grounds. Houston provides the changes in the shadow value of water when the quantity of water is reduced by 10% in the Yakima and Umatilla regions.

Estimates of the Impacts of New Irrigation Projects

In addition to the economic studies on the value of irrigation water, other research related to opportunity cost measurement involves the evaluation of future irrigation development. The opportunity cost of water reallocated to agriculture is reflected in the value of the lost hydropower, rather than in a dollar value unit. Two of these studies are very briefly noted in this section. The Whittlesey (1980), the Butcher et al (1986), and the McCarl and Ross (1985) studies discussed above also fall into this category.

Corps of Engineers (1976)

The thrust of this study was to identify and evaluate the impacts of alternative future irrigation development levels and minimum instream flow rates. Two scenarios were considered: an increase in the amount of irrigated land by 3 million acres; and an increase of 4.2 million acres. They estimated that 770 MW of power would be lost under the first scenario, and approximated 966 MW under the second scenario. These estimates were obtained from the Corps of Engineers' Hydro System Seasonal Regulation computer program, and did not reflect the increased amounts of power that would be used in transporting and applying irrigation water.

Davis (1979)

The purpose of the Davis study was to establish a baseline projection of the impact of existing and projected new irrigation development on the hydropower system in the region. The baseline assumed that current (1975?) transportation and application efficiencies and water and energy use rates would prevail. The projected increase in irrigation development was approximately 4 million acres in Oregon, Washington, and Idaho. The findings of this study were that the entire irrigation activities in the region (which were approximately 11.4 million acres including the projected 4 million new acres) would result in a total depletion of 22 million acre-feet. The forgone hydropower would be 2,032 MW and the pumping and transportation would consume another 1,176 MW.

APPLICATION TO THE YAKIMA AND UMATILLA

This section is an example of a pilot application of the pricing information reviewed above to the Yakima and Umatilla subbasins. This sample application is performed in order to demonstrate a trial usage of existing studies as they might be applied to two areas of the Columbia River Basin that have been identified by the Northwest Power Planning Council as having particularly acute conflicts between irrigation withdrawals and fish passage and production. We wish to emphasize that this is only intended to be a illustrative example of how previous work could be used; it is definitely not meant as the final word on how such an approach would work in practice. As noted in the introduction, this type of analysis explicitly assumes the existence of water rights markets in the region, and further assumes that water rights could be purchased with the express intention of not using the water for irrigation, but instead using it to assist in restoration of the salmon runs. This exercise is not meant to advocate the establishment of markets in irrigation withdrawal rights, but simply to point the way toward how an opportunity cost analysis might proceed.

The Yakima basin water supply problems are explained in detail in Section (803)(a) of the Council's 1987 Fish and Wildlife Program. Briefly, however, they can be described as follows. The demand for irrigation water in the Yakima is a substantial portion of the subbasin's natural flows, and the demand is concentrated in periods when those flows are very low. Although the subbasin already has a number of storage reservoirs, the Council has proposed that additional storage facilities be constructed in order to augment instream flows to benefit the salmon and steelhead stocks that spawn in the area. The Council has also urged responsible parties in the area to enhance existing water conservation efforts and perhaps to undertake some new measures, such as pressurized delivery systems instead of open canals. In the Umatilla, the Council has proposed that flows in the Umatilla be augmented with water pumped directly from the Columbia River, with the objective of enhancing flows for fish (see Section (703)(a)(17) of the 1987 Program). Although the Council is clearly sensitive to possibilities for decreasing the demand for irrigation water in both subbasins, there are limits to what they can accomplish on the demand side under the terms of the Regional Act, and the proposals to increase water supply will be costly to implement. This suggests (at least to economists) that market alternatives. and in particular water rights markets, may be worth investigating.

Even taking water rights markets as "given", several additional assumptions are required in order to perform the pilot analysis. First, since both the Yakima and Umatilla supply augmentation programs are still in the planning stages, it will be necessary to assume some quantities of augmentation, or conversely, to assume quantities for reductions in irrigation withdrawals that are ecologically symmetric with augmented flows. Second, one would ideally like to have a marginal willingness-to-pay schedule for each subbasin "mapped out" in small increments over a wide range of proposed changes in flows. Instead, the data presently available are point estimates of the marginal productivity of water that apply over broad ranges of withdrawals. It should also be noted that by reducing withdrawals, there might be additional benefits to the fish populations, especially since return flows, which generally are of low quality, would also be reduced. In addition, there may be effects on regional commodity prices, if crop production patterns changed markedly as a result of the decrease in irrigation. Finally, regional electricity consumption would decrease to the extent that the water rights being purchased apply to water that is pumped with electricity. Therefore, this can only be considered an illustrative example of the type of analysis that might eventually be performed if better pricing information were available.

Houston and Whittlesey (1985, p. 52) report that the Yakima basin diverted about 3 million acre-feet of irrigation water in 1982 (surface water diversion), and that about 430,000 acre-feet of surface water were diverted in the Umatilla that same year. The questions that one would like to answer are: at what price would water rights change hands; and what would be the total dollar value of the exchange. The motivation for the exchange from the purchaser's viewpoint would, of course, be to "save" the water to enhance flows for fish, with water rights presumably being purchased by some regional authority. The motivation from the seller's end would be to make more money by selling the water than by applying it to crops. While detailed pricing information is not presently available, some "flavor" for what the results might be can be gained from an analysis of Table 5.6. The table contains the results of Houston and Whittlesey's analysis for water rationing, showing shadow prices for incremental percentage reduction in water rights. If, for example, one were to reduce surface water withdrawals by 5 percent in the

Yakima, the marginal value of the last acre-inch of water not used for irrigation would be $4.27. In the Umatilla, the marginal value for the same percentage reduction would be somewhat higher, at $7.15 per acre-inch. Note that these numbers are only proxies the marginal price of reducing total withdrawals via some sort of rights purchase mechanism. They represent the minimum prices for which a farm would sell its rights, not the price at which the sale would actually take place. Also, it appears from the study that these shadow prices are derived by considering subbasins as homogenous units; a more detailed analysis would probably produce different shadow prices.

Having made these many assumptions, the arithmetic turns out to be quite straight-forward. Assume that the aim is to increase flows or decrease withdrawals on the Yakima by 300,000 acre-feet per year. What would purchase of rights cost in order to accomplish this? From Table 5.6 we find that the shadow price per acre-inch of water at 5 percent reduction is $4.27 per acre-inch, and for a 10 percent reduction it is also $4.27. The goal of 300,000 acre-feet is about 10 percent of the annual withdrawals, so the annual cost of rights purchase would be :

(300,000 acre-feet) * ($4.27 per acre-inch) * (12 inches per foot)

or approximately $15.3 million per year. A similar calculation is shown for the Umatilla, assuming a 10 percent purchase (of the 430,000 acre-feet per year withdrawn in 1982) and that the same price (i.e., the shadow price for 10 percent reduction) is paid for all rights purchased:

(43,000 acre-feet) * ($7.35 per acre-inch) * (12 inches per foot)
or about $3.8 million per year.

These examples are extended in Table 5.9 for different percentage reductions from base case water withdrawals.

These are simple, illustrative examples of how the pricing information from previous work could be applied to the Columbia. While these values

Table 5.9. Illustrative Costs of Irrigation Trades for the Yakima and Umatilla

Reduction Scenario[a]	Yakima			Umatilla		
	Amount Purchased (AF/Yr.)	Marginal cost ($/AF)	Total Cost[b] ($/Yr.)	Amount Purchased (AF/Yr.)	Marginal cost ($/AF)	Total Cost[b] ($/Yr.)
5 %	150,000	$51.24	$7,686,000	21,500	$85.80	$1,844,700
10 %	300,000	$51.24	$15,372,000	43,000	$88.20	$3,792,600
15 %	450,000	$68.40	$30,780,000	64,500	$124.20	$8,010,900
20 %	600,000	$68.40	$41,040,000	86,000	$124.20	$10,681,200

Notes:

a. The annual base case withdrawals for the Yakima and Umatilla are assumed to be 3,000,000 AF and 430,000 AF, respectively.

b. Total Cost is annual amount purchased multiplied by the marginal cost. This assumes that the purchaser is not a monopsonist, and cannot discriminate among perspective sellers.

appear high, this may be due to the nature of the crops grown in the subbasins. To obtain more accurate estimates of actual costs and how such a market might really operate will require considerable additional effort.

INSTITUTIONAL BARRIERS AND POTENTIAL BENEFITS OF IRRIGATION RIGHTS MARKETS

This section is concerned with the institutional barriers to trading in irrigation rights, and with the potential benefits of such trades if they were permit ted. In economic terms, the barriers can be thought of as prohibitively high transaction costs. The potential benefits are Pareto-improvements for both present irrigators and for government agencies, including BPA, that have responsibilities for fish mitigation activities. The remarks that follow are meant to apply to the entire basin, and not just to the Yakima and Umatilla.

Before discussing the problems with instituting water rights markets in the Columbia basin, it should be pointed out that there are some limited areas in the West where water rights are freely traded, both among irrigators and between irrigators and other water users (see Frederick and Kneese, 1988, for details). In the context of fish mitigation, for example, a trade was recently arranged between irrigation and wildlife management agencies in the Upper Colorado (Water Market Update, 1988). The regions with active rights markets seem to have several characteristics that differentiate them from the Pacific Northwest. The first is that they are primarily located in areas where surface water has always been a scarce resource. This is in contrast to the Pacific Northwest, where perceived scarcity is a relatively recent phenomenon. The second is that state or regional irrigation authorities were empower-ed to supervise and administer both water distribution and trades at about the same time that the water became available for use by agriculture. Finally, ownership of water- potentially available for trading is generally much more clear--cut in areas wi th ac tive markets, and, at least in some cases, is separable from the land to which the water has traditionally been applied.

There are several classes of institutional barriers to Northwest regional markets in water rights. These are discussed in detail in Butcher et al. (1986), in the context of expanding the region's irrigated acreage, and are not repeated here. Instead, we attempt to outline the major potential problems with setting up water rights markets. This discussion will be confined to fairly general legal and institutional considerations, and will not touch on particular laws and agencies related to regional water rights. The context for the discussion is the reduction of irrigation withdrawals in order to increase instream flows for the purpose of mitigating impacts on the anadromous fish populations. It is assumed that irrigators would only participate in such a market if it would benefit them economically, by producing more income from selling water rights than could be gained by using the water for irrigation.

In order to understand the institutional restrictions on trading, it is useful to consider the background of irrigation development in the region. Up until the Great Depression, there were few large-scale public water works in the region, and most irrigation was conducted on a small scale, by individual farmers or small groups constructing relatively primitive diversion structures and conveying water to fields via simple gravity-fed systems. Storage capacity was very small relative to annual flows, and so any water not diverted was in effect wasted, in so far as consumptive uses of the water were concerned. Under these conditions, the following doctrines were developed. First, in-stream use of water was subordinated to "appropriative" use for irrigation. Second, where formal allocation of irrigation water was required, the "senior" irrigator(s) who had been the first to divert the water were considered to have rights that superseded those of "junior" irrigators, who had not begun their diversion activities until after the senior diverters had their rights established. The irrigators were required to put the water to "beneficial" use, which in practice simply meant that it was to be used to grow crops, and not wasted indiscriminately. So long as irrigators did apply their water to beneficial use, they were effectively entitled to as much water as they could physically obtain, subject to the junior/senior restrictions. The intent of the beneficial-use provision was to prevent withdrawals that simply interfered with others' use of the water. While the irrigators were loosely regulated by a variety of state and local institutions, they were not very strictly supervised so long as abundant steamflow was available.

These policies, while certainly not promoting efficient use of the resource, were not a serious problem so long as withdrawals comprised a modest portion of flows (at least under normal flow conditions), and so long as there was little or no demand for instream uses of the rivers. Problems began to arise as withdrawals increased relative to flows, and as demands for instream use of the water increased. In the Columbia basin, three developments lay behind the increased consumptive and instream demands. First, with the completion of the Grand Coulee project, irrigators could withdraw much more water, and pump it to much greater heights. This trend continued with the completion of other hydro projects in the basin. Second, as hydropower and storage project development continued, much less of the spring and early summer flows were left in the river, as much of it could be stored for use in in hydropower production later in the year. Finally, with the passage of the Regional Act in 1980, fish and wildlife are to be accorded equitable treatment with power interests, which indirectly increases the need for instream flows.

This combination has made water a scarce resource, in the economic sense of the term. Instream flows are needed both by irrigation, hydropower, and fisheries management interests; and flows are at present fully allocated. Therefore, in order to increase instream flows for fish habitat and passage, flows must be re-allocated from irrigation, hydropower, or both.

Unfortunately, although the research reviewed above on the economic value of irrigation water indicates that both irrigators, fisheries, and hydropower interests could be made better off as a result of marketing irrigation water rights, there are formidable institutional barriers to such an activity. The barriers are a result of the relatively abundant streamflows that prevailed at the time that irrigation laws and management agencies were instituted, and the emphasis on providing irrigation water supplies as inexpensively as possible, from the viewpoint of the individual irrigators. These factors have resulted in both poorly defined property rights and high transaction costs.

The property rights to water are unclear because of the “use it or lose it” nature of the withdrawal rights. Unless the water is put to beneficial use, rights to the withdrawal can be lost. Under these circumstances, it is almost impossible for an irrigator to lease or sell their rights to another party. In addition, third-party interests must be protected in any re-allocation, usually so that downstream users of return flows will still have water available to them. Establishing that these interests are protected in a trade is, in practice, almost impossible in areas where there are no agencies designed to manage trading, and results in prohibitively high transaction costs. As noted above, however, there are areas in the West (especially in New Mexico and Colorado) where state agencies take an active role in facilitating trade or sale of water rights and in protection of third-party interests. Finally, the legal status of trades between states and between basins is problematic, and state agencies in the region are not equipped to administer trades in any case.

The potential benefits from trades are high, however, and may justify the political and economic costs of modifying the institutional and legal apparatus needed to make trading and sale of irrigation rights practical. The economic benefits include increasing income for irrigators and reducing both costs of flow modification for fish and the costs of hydropower.

The income to irrigators could be increased because, under present arrangements, farmers in many parts of the Columbia Basin are using large amounts of water to produce relatively low-valued CL-ops, including pasturage, wheat, and field corn. Assuming that a “grandfather” clause allowed present irrigators the right to their past water allocations for free, preliminary evidence suggests that they might well be able to market their water rights for considerably more than the net value of of the crops produced using the water. Thus, many farms might well make more money by selling water rights than by selling crops.

For an agency or authority with responsibility for fish mitigation activities, purchase of rights may well be the least expensive method for augmenting flows for fish habitat and passage improvement. The alternatives to rights purchase, including construction of additional storage reservoirs and pumping water from larger rivers uphill to smaller tributaries, tend to have high capital costs. If at some future time it turns out that flow augmentation is not the best way to mitigate fishery impacts, the capital costs cannot be recovered. In contrast, short-term leases of water rights would be a much more flexible management alternative for a mitigation agency.

The hydropower production of the basin could also be enhanced substantially using irrigation rights purchases. While this is obvious to even the most casual observation, a somewhat more subtle point is that firm power production could also be increased (Whittlesey et al., 1986). If this were institutionally feasible, then the region's use of expensive thermal power sources could probably be reduced significantly, resulting in lower electricity prices for consumers.

Quantifying these benefits is not easy, of course. While attempts have been made in the literature cited above to do this on a case-by-case basis, no effort has been made to do so in the requisite detail for the basin as a whole. While such an effort would be a major undertaking, it is, for the most part, a straight-forward extension of work that has already been done by Houston, Whittlesey, and others. One important exception to this generalization is that if water were re-allocated on a large scale, the operation of the hydrosystem would need to be re-optimized in order to account for the additional water available for power generation. In addition to the empirical work, additional research would be required on how the present institutional and legal framework could be changed to accomodate trades and sales of water rights. Al though these are very sensi tive political issues, the potential benefits to all parties concerned suggests that they may well be worth pursuing.

TOWARDS EMPIRICAL IMPLEMENTATION

In this chapter we have focused on methods for quantifying the opportunity cost of water allocated to fish enhancement activities. In doing so we have first specified a framework which minimizes the cost of achieving some desired level of fish runs, and then proceeded to suggest methodologies to measure the shadow values of water in the two major market uses of water from the Columbia River system, irrigated agriculture and hydropower. Finally, we have provided a brief summary of empirical research on the economic value of irrigation water and on a related issue, the tradeoff in the use of stream flows for irrigation development or hydropower generation.

The opportunity cost of water is defined in terms of the returns at the margin in its alternative uses. The basic idea is that the cost of any strategy for reallocation of scarce instream flows can be measured by the value of the best alternative that must be foregone. The opportunity cost concept is particularly useful in clarifying the alternatives inherent in making choices which involve use of limited resources. The range of alternative uses of a resource will depend upon the time period allowed for adjustment or the time frame of the analysis; in the short run, alternatives may be quite limited, and as a result the opportunity costs are relatively small.

More specifically, the previous discussion has suggested two approaches to quantifying the opportunity cost both of which are based on neoclassical production analysis. The first requires estimation of the long run production function, or its dual cost function. This would require information on the cost of adopting the alternative technology, and the effects on long run profit maximizing levels of the other inputs. This type of analysis could be done using engineering-econometric approaches. The second approach was based on the used of restricted cost and profit function models. This willingness to pay function or the shadow value for the surface water is used to measure the opportunity cost of the resource to the agricultural producers.

One method to obtain information on the willingness to pay would be to use a survey method which simply asks what these functions are. A willingness to pay survey would need to be very carefully designed, in order to elicit accurate estimates of farm owners' real demand functions. A second method would be to utilize econometric techniques to estimate the parameters for the functional forms for the restricted profit or restricted cost function. The functional forms chosen should be flexible, providing a second order approximation to any arbitrary twice continuously differentiable function. Examples include the translog and generalized Leontief functional forms. The econometric model would consist of the profit or cost function and the set of share equations for the variable inputs and outputs. This approach will provide an indication of the statistical significance of the opportunity cost estimates. Econometric estimation of these temporary equilibrium models appears to be the most promising means of obtaining an proxy for the shadow value of surface water to irrigated agriculture in the Pacific Northwest.

To estimate these short run production models one would need pooled cross section-time series data on the prices and quantities of the outputs and inputs in each major irrigated area. In the past, the Census of Agriculture data has been used for many of the econometric studies as noted above. However, since 1969, this data set no longer provides information on the amount of irrigation. In order to utilize the Census of Agriculture data one would need to collect additional data on the irrigation variables in such a way that it matched up with the remaining Census information. Some of these data may already exist at various universities (Oregon State University, Washington State University, University of Idaho) and could be supplemented by additional surveys. Initial discussions with various researchers at these institutions indicated that such information would require additional "legwork", but it was probably a feasible undertaking.

We would also need information on the degree to which the crops grown in these areas are government supported crops. With the exception of wheat, the other major crops,-- apples, vegetables, potatoes, and alfalfa-- are not directly affected by support or target prices. As a result, the agricultural prices for these latter crops may adequately reflect market conditions. For wheat, one would want to adjust the reported prices for the government induced distortions in the years when the commodity programs were binding. Information on the level of government supported crops, by state, is readily available from the USDA.

A second source of information would be crop budgets. Most of the programming models which estimated an economic value for irrigation water relied on these budgets, which are generated by the Cooperative Extension Services in Washington, Oregon, and Idaho. For these budgets, the production practices, costs, and product prices are based on a specific cropping season or indexed to a particular year. A fairly detailed discussion of the crop budgets can be found in Houston and Whittlesey (1985, pp. 37-41).

If one is satisfied with the use of programming models and the associated crop budgets to provide adequate measures of opportunity costs, then a fairly direct means for further development of empirical estimates would be to extend and update the Houston-Whittlesey model. This is not quite as simple as it sounds, since it would first involve updating the production area farm models and crop production activities, and then updating the production cost budgets. Finally, one would want to adjust the constraints in the model to reflect the resource constraints as well as the modified production area constraints.

It appears that studies involving further refinement of water response relationships and deficit irrigation, and intraseasonal water- scheduling, as well as allowing for long run alternative irrigation technologies would better reflect the role of irrigation in the region's agricultural sectors, and provide better estimates of the economic value of the irrigation water.

For estimating the opportunity cost of water used in the hydropower sector, the methods are even less direct. The publicly regulated rate structure for pricing electricity and the current excess capacity in power generation obscure the connection between observed prices and opportunity costs. Virtually all electric utilities base their pricing on average costs. Since these rates do not reflect the marginal cost of production, the opportunity cost cannot be determined solely from market prices.

The conventional approach to assessing the opportunity cost of the lost streamflows has been to use the replacement cost methods. This amounts to evaluating the foregone output in terms of the costs of the alternative (thermal) plant that would be required in the system to replace the hydropower capacity. This is a valid means of evaluating the opportunity costs if it has already been determined that the output should be produced.

The replacement cost appears to be less justified during periods of excess capacity. Under these conditions one needs to value the lost output in terms of its social value, which will vary annually depending on the annual flow levels. Given the current surplus of firm and nonfirm energy in the region, the social value of the lost hydropower may be equivalent to the potential net revenues from sales outside the region.

This suggests that quantifying the opportunity costs of water reallocated from hydropower will require a careful scrutiny of the flow levels and the market conditions at the time. The latter information may be contained in the electricity demand projections which are performed by the public utilities. To simply utilize a single rate, such as the reported replacement cost for firm power from a thermal plant, is likely to overstate the true social cost of the streamflows diverted from this sector.

Chapter 6

Data Needs for Proposed Pilot Study

INTRODUCTION

This chapter describes the data that would be needed for a pilot study of a subbasin. These data will be used to develop the economic-ecologic simulation model outlined in chapter 2 and the least-cost model described in chapter 3.

Data needs for the development of the economic-ecologic simulation and least-cost models are organized in four groups.

- Hydrosystem Data
- Fish Life-Cycle Data
- Direct Costs of Fish Mitigation Alternatives
- Opportunity Costs of Fish Mitigation Alternatives

The first group of data is needed to describe the physical characteristics and operations of the hydrosystem. This includes the unregulated streamflows, the locations of dams and other hydraulic structures, and the physical characteristics of dams such as the sizes and hydraulic capacities of turbines and generators and the design of adult fish ladders. The second group is needed to describe the life-cycle of the anadromous fishery. This includes data that describe fish biology and data that describe fish habitat other than streamflow. Examples include the quantity and quality of spawning and rearing habitat, the production of smolts from eggs, and the losses of smolts in reservoirs, through turbines and by-pass facilities, and over spillways. The third group is needed to estimate the direct costs of alternative fish mitigation measures. The fourth group is needed to estimate the opportunity costs of fish mitigation measures and procedures. This group includes data that are needed to estimate the opportunity costs of reductions in the generation of hydropower and of potential reductions in withdrawals of irrigation water (through purchases of water rights), for the benefit of the anadromous fishery.

The data needs are organized in these four groups because they involve different disciplines, research methods and approaches, teams of researchers, and organizations responsible for the collection of these data. Moreover, these four groups divide conveniently along the lines of the major components of the economic-ecologic simulation model described in chapter 2 and shown schematically in Figure 2.1. They are also consistent with the hydrosystem and fish life-cycle modules that comprise the least-cost model described in chapter 3. Finally, data collection and model development in these four areas can proceed somewhat independently of one another, at least in the beginning, as long as the overall framework for the economic-ecologic simulation model has been developed, including the locations (in the simulation model) of the fish mitigation measures, the inputs to and outputs from each of the major components of the simulation model, and the overall development of the simulation model. The data that are needed for the development of the least-cost model are the same data that are needed for the development of the economic-ecologic simulation model.

The data needs described in this chapter are generic in the sense that they do not pertain to a particular subbasin or to particular fish mitigation measures or procedures. They are general data needs that would apply to any, or all, of the subbasins in the Columbia River Basin. There are two principal reasons for keeping the discussion of data needs as general as possible at this point. The first reason, as described above, is that the subbasin for a possible pilot study has not yet been selected. The second reason is that much of the data needed for model development will come from the subbasin studies under the supervision of the Northwest Power Planning Council and currently being conducted by the agencies and tribes. The data needs described in this chapter should be viewed as preliminary.

HYDROSYSTEM DATA

The hydrosystem data include the data that are needed to describe both the physical characteristics of the hydrosystem and the operations of the hydrosystem. Specifically, these data include the unregulated streamflows, the physical characteristics of the river system, the physical characteristics of dams and reservoirs, the operating characteristics of dams and reservoirs, hydropower production, irrigation water withdrawals, and fish mitigation measures and procedures that pertain directly to the operation of the hydrosystem. The hydrosystem data do not include biological data on fish, or data on fish mitigation measures that do not pertain to the operation of the hydrosystem such as hatcheries and improvements in natural spawning and rearing habitat. These data are included in the fish life-cycle data discussed below.

Unregulated Streamflows

The first kind of hydrosystem data needed for model development is the record of the historic unregulated streamflows in the basin. The primary source of these data is a report prepared by the Depletions Task Force, “1980 Level Modified Streamflow, 1928-1978” (Depletions Task Force, 1983). This report is the result of the Depletions Task Force’s efforts to reconstruct the unregulated flows in the basin in the absence of hydro projects, after allowing for irrigation water withdrawals, evaporation, and other factors that affect water balances. There is no completely unambiguous way to do this, since the flows in the basin have been heavily regulated for much of this century. Nonetheless, the unregulated streamflows produced by the Depletions Task Force seem to be accepted in the region as a reasonable approximation to the natural inflows for the fifty-year historic period. 1928-1978. These data would be used to develop the different energy rule curves (described in chapter 2) that are needed to construct an operating r-ule curve for each of the storage projects; also, to reconstruct the regulated streamflows in order to assess the impacts of changes in these flows on the anadromous fishery.

Physical Characteristics of the River System

The second kind of hydrosystem data needed for model development describes the river system. This includes the locations of major rivers, tributaries, significant water withdrawals, dams and reservoirs, and other structures and developments that affect regulated streamflows. These physical attributes of the hydrosystem define the “nodes” in the economic-ecologic simulation model described in chapter 2.

Physical Characteristics of Dams and Reservoirs

The third kind of hydrosystem data needed for model development pertains to the physical characteristics of the dams, reservoirs, and other hydraulic structures in the basin that affect streamflows or fish passage, or both. These data include the physical dimensions of dams, the sizes and capacities of turbines and generators, the capacities and effectiveness of by-pass facilities, the capacities of spillways, and the storage-level relationships for both the storage dams and the run-of-the-river dams. The source of data for the Corps of Engineers projects is the District Office of the Corps of Engineers in Portland (Corps of Engineers, 1985). The sources of data for the private utility dams are the private utilities.

Operations of Dams and Reservoirs

The fourth kind of hydrosystem data needed pertains to the operation of dams and reservoirs. These data include the fixed (upper and lower) rule curves for the storage reservoirs, the various energy rule curves (generated by the Systems Analysis Model and the BPA Regulator described in chapter 2), and the proportion of (monthly) flows passing through the turbine, through the by-pass facilities, and over the spillway.

Hydroelectric Production

The fifth kind of hydrosystem data needed for model development pertains to the demand--more accurately requirements--for hydroelectric power. Two kinds of hydropower requirements need to be distinguished. The first requirement concerns the firm, or primary, hydropower. The second requirement involves the surplus, or secondary, hydropower. These two hydropower requirements are treated differently both in the analysis of fish mitigation alternatives and in the measurement of the opportunity costs of reductions in the generation of hydropower for the benefit of the anadromous fishery (discussed below).

Irrigation Water Withdrawals

The sixth kind of hydrosystem data needed pertains to water withdrawals for irrigation. These water withdrawals can have a significant impact on streamflows, and thus on the fisheries. Water withdrawals for irrigation reduce streamflows below the locations where the water is withdrawn. They also consume significant quantities of hydroelectric power in those situations where electricity is used to operate the irrigation pumps. Return flows from irrigation may supplement streamflows below the irrigated acreage.

For the analysis of the opportunity costs of reducing withdrawals of irrigation water (through water rights purchase) for the benefit of the anadromous fisheries (described below), additional data on the types and quantities of crops grown, the prices of those crops sold in the market, and the effects on crop production of reductions in the application of irrigation water will be required. This is discussed in more detail in chapter 5 and in the section below on data needs for estimating opportunity costs.

Fish Mitigation Measures

The last kind of hydrosystem data needed pertains to fish mitigation measures and procedures that involve physical changes to the hydrosystem and changes in the operation of the hydrosystem for the benefit of the anadromous fishery. Physical changes include by-pass facilities for the downstream passage of juvenile fish, and fish ladders for the upstream passage of adult fish. Changes in the operation of the hydrosystem include altering timing of releases from storage dams, reduced flows through the turbines and increased flows over spillways.

FISH LIFE-CYCLE DATA

The fish life-cycle data include data needed to develop the fish life-cycle simulation model described in part I and data needed to describe the fish mitigation measures and procedures that pertain to the fisheries but not to the operation of the hydrosystem. Examples of fish mitigation measures in the latter category include fish hatcheries, improvements in natural spawning and rearing habitat, and the transportation of smolts by truck and barge to below Bonneville Dam.

Fish Life-Cycle Simulation Model

The fish life-cycle simulation model describes the life-cycle of the anadromous fisheries from the laying and hatching of eggs in hatcheries and upstream tributaries, through the raising of fry, the migration of smolts down the Columbia to the ocean, and the ocean fishery, to the return of adult fish to hatcheries and natural spawning areas. This simulation model also includes those fish mitigation measures and procedures that do not involve either the physical alteration or the operations of the storage and run-of-the-river dams. The fish life-cycle simulation model has a monthly time step (actually 14 periods per year to be consistent with the hydrosystem simulation model), and it will be run for a planning period of 15, 20, or more years. The fish life-cycle simulation model will be developed as part of the research described in part I. Therefore, the description of the data needs for the fish life-cycle simulation model in this chapter is brief because it duplicates descriptions of data needs in part I.

There are five principal stages in the life-cycle of anadromous fish. They include smolt production, downstream migration of smolts, estuary and early ocean survival, late ocean survival and ocean harvest, and upstream passage of adult fish. The data source for all stages would be parts I and III of this report.

DIRECT COSTS OF FISH MITIGATION

The direct costs of fish mitigation measures and procedures include the capital, operating and maintenance, and land costs of physical alternatives installed to increase the production of adult fish. These include alterations to dams such as by-pass facilities for juvenile fish and fish ladders for adult fish, transportation of smolts around dams to below Bonneville Dam, the construction of new and the expansion of existing fish hatcheries, and physical improvements in natural spawning and rearing habitats. Estimates of the direct costs of fish mitigation alternatives will be obtained from the Corps of Engineers, the Northwest Power Planning Council, private utilities, and the subbasin planning efforts currently underway in the basin.

The desired output of this part of the research is the development of cost functions that relate levels of fish mitigation to the costs of achieving those levels. These cost functions will be developed from data on the effectiveness of particular mitigation measures and the costs of those measures. An example is a cost function that represents the capital, operating and maintenance, and land costs of constructing a fish hatcheries of different capacities (number of fry produced) at a particular site. Another example is a cost function that represents the costs of developing and maintaining different quantities of natural spawning or rearing habitat, in terms of the number of fry or smolts that can be supported.

OPPORTUNITY COSTS OF FISH MITIGATION

The opportunity costs of fish mitigation measures and procedures are losses to the region (or to society more generally) of changes in the operation of the hydrosystem for the benefit of the anadromous fishery.

These losses are not expenditures per se, but rather reductions in benefits to other activities.

There are two kinds of opportunity costs relevant to the cost-effectiveness analyses described in this part (chapter 1). The first kind of opportunity cost is the loss in the production of hydropower due to diversions of flows through by-pass facilities and over spillways for the benefit of the anadromous fishery. The water budgets provided the fishery at Priest Rapids and Lower Granite dams between between 15 April and 15 June each year is an example of this kind of loss (Northwest Power Planning Council, 1987a). The second kind of opportunity cost involves the loss to agriculture due to reductions in withdrawals of irrigation water, for the benefit of the anadromous fishery.

Opportunity costs are not easy to measure. The market data needed to measure opportunity costs directly are seldom available. The prices of water rights could be used if there were a market in water rights and if prices had already been established. Otherwise, measurement of the opportunity costs of fish mitigation alternatives is more of a research effort than a data collection effort, as described in chapter 5.

Opportunity Costs to Hydropower. The use of the Systems Analysis Model (SAM) to generate the opportunity costs of reductions in the production of hydroelectric power would need to be explored. For the most part, the economic losses generated by SAM in this context represent the costs of additions to the region's thermal electric generating capacity to compensate for reductions in the production of hydroelectric power to improve fish passage.

Potential sources of data for the opportunity costs of reductions in the production of hydroelectric power are discussed in chapter 5. It is our tentative conclusion that considerable additional research needs to be undertaken in this area to develop more accurate estimates of the opportunity costs of modifying hydropower generation patterns to meet the region's goal of increasing adult fish populations. Most of the work that has been done on the opportunity costs of hydroelectric power uses the costs of constructing and operating thermal power plants as a “proxy” for the real opportunity costs. However, this approach may overstate the opportunity costs that would actually be realized over the long run because of energy conservation and other alternatives for reducing the demand for electric power generated in the region.

Opportunity Costs to Agriculture. The opportunity costs of reductions in withdrawals of irrigation water could be estimated from existing studies, as described in chapter 5, or from additional research to be conducted in the next phase of the research. Our tentative conclusion is that while existing studies clearly provide an excellent starting point for future investigations, more detailed information on the losses to agriculture due to reductions in irrigation water will be needed. Another possibility for measuring the opportunity costs of reductions in irrigation water withdrawals would be to use data from water rights markets, if they come into existence in the region. This is a more direct approach to estimating the opportunity costs as it uses the prices of irrigation water directly rather than imputing the value of irrigation water from the prices of agricultural crops sold in the market.

CONCLUDING COMMENTS

Additional work on the cost-effectiveness problem would require both data collection and research. For the hydrosystem portion of model development, most of the effort would involve the collection of existing hydrosystem data. For the fishery life-cycle portion of model development, the effort would be equally divided between data collection and research. This is discussed in more detail in part I. Some of the direct cost information is already available and other direct cost information is under development in the region. This information would need to be gathered, organized, and analyzed in the next phase of the research. In addition to the data that are (or will be) available, some of the direct costs of fish mitigation would need to be estimated and all the cost functions for the different fish mitigation measures and procedures will need to be developed.

The opportunity costs of reductions in the generation of hydroelectric power and of reductions in withdrawals of water for irrigation would require additional research to build on existing studies. In the case of irrigation, this would primarily consist of expanding upon previous work while for hydropower more fundamental research will be required.

In addition to the development of the hydrosystem simulation model, the fishery life-cycle simulation model, the cost functions for the direct costs of fish mitigation, and the cost functions for the opportunity costs of fish mitigation, a major research effort would have to be devoted to developing the overall computational framework for the economic-ecologic simulation model described in chapter 2, to integrating the different components of that model, and to developing the system-wide least-cost model described in chapters 3 and 4.

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Appendix to Volume 2 Part II
Possible Approaches to Allocation of
Joint Costs of Fish Mitigation*

INTRODUCTION

This appendix addresses task 2 of the BPA/RFF work statement, allocating costs and responsibility for the loss in fish productivity. The task specifically states that RFF was to

"design a study to assess alternative procedures for allocating responsibility for loss in fish productivity:

a) to the hydroelectric purpose of federal hydroprojects,
b) between federal and non-federal hydroelectric projects, and
c) to systemwide loss caused by hydroelectric system development and operation, but not attributable to project(s) of any single owner."

Among other possible purposes, loss allocation forms the basis for an allocation of fish mitigation costs as required by the Regional Act. As with the other tasks, the emphasis here is on design of research, not on research itself. Therefore, the methods outlined here should be taken as preliminary guides to future Phase III work, and not as the final word on the subject.

Cost allocation is a common problem in resource economics, particularly when dealing with large, multi-purpose projects. As a general matter, joint costs arise in investment projects and in the daily operations of private firms and public agencies because of what has been termed economies of scale

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*This material is included as an appendix to Part II of volume 2 instead of a part of the volume proper for two reasons: (1) it has not undergone the public review that the other sections have, and (2) it presents some conceptual issues it was not possible to resolve in the Phase II research and which must await further consideration in Phase III.

in activities. That is, adding a use or user to an investment project often does not increase total cost by as much as the cost of a stand alone facility built to serve that use or user. But, at the same time, the sum of the "separable costs," those identifiable costs of adding each use or user, is smaller than the total cost of the multi-user project, and the joint costs must then be allocated to the individual users or purposes.

The 1980 Regional Act (16 U.S.C. $839) places some constraints on the fish mitigation cost allocation process. In particular, it states that "consumers of electric power shall bear the cost of measures designed to deal with adverse impacts caused by the development and operation of electric power facilities and programs only" (Section 4(h)(l)(B)). Furthermore, Section 4(h)(l)(C) states that "the amounts expended . . . shall be allocated ... among the various hydroelectric projects of the Federal Columbia River Power System. Amounts so allocated shall be allocated to the various project purposes in accordance with existing accounting procedures . .." As will be shown below, these legal constraints on the allocation of joint costs assume considerable importance in the choice of allocation methods.

In the classic cases the origins of this jointness lie in the production technology, where that term is defined broadly enough to encompass such diverse examples as:

- a water resource investment project involving construction of a dam to produce water storage for flood control, flat-water recreation and navigation, and storage and usable head for hydro power;
- a satellite system providing capacity for telephone, television, and data transmission over long distances.

In such cases, however we attribute costs to a use, be it via econometrically estimated marginal cost functions, or engineering – derived separable cost calculations, <u>the result is intrinsically arbitrary</u> because the capital goods involved serve more than one use simultaneously. The dam holds back a pond at a particular elevation, and the water serves recreation and navigation while it is ponded and hydropower generation when it is released. The satellite, with its attendant launch costs, serves all transmission uses by being in place, though particular parts are clearly assignable to particular uses. In such cases there is no non-arbitrary way to divide total costs among the uses. There are, however, a number of competing ways of structuring the arbitrary process, and these may be distinguished on the basis of a number of characteristics they display when confronted with particular problems.

CHARACTERISTICS OF COST ALLOCATION METHODS

The characteristics of cost allocation are discussed here in terms of game theory. Game theory is frequently used by economists and others to analyze cost allocation problems. It is a field of inquiry on the borders of several social science disciplines, and involves analysis of outcomes of strategic interaction between players who jointly influence the outcomes that each receives, and know it. Although its applicability to the present problem, which is essentially one of cost accounting, is somewhat limited, it still provides a useful framework for discussing the problem and especially for formulating it mathematically. See R.D. Luce and H. Raiffa, *Games and Decisions* (New York, Wiley: 1958) for an early but still useful survey.

Ex-ante versus Ex-post Allocation

One important characteristic of the present problem is that jointness arises in a different way and at a different stage than is true in the classical examples. Usually, economists are interested in allocating costs of a project that is in the planning stages (i.e., *ex ante*). In the case of fish mitigation, however, the costs are to be decided on and borne after the rest of the capital projects, namely the Columbia River dams, are in place, (i.e., *ex post*) . The fish mitigation purpose is not a use of the dams in the same sense as is hydro power. Rather it is a “correction” to the original uses. One way of viewing it, therefore, is as an *ex-post* cost overrun; a point of view that has significant implications for the results with respect to methods, especially if the goal is ex-post accounting.

Joint cost allocation as an exercise with other than ex-post accounting goals, usually involves ex-ante cost estimates for various hypothetical single and multiple purpose projects. On an ex-ante basis, prospective charges can be estimated for each use or user and commitments can be sought from users free to buy in or not--or decisions can be made by a central decision-maker about whether to include or not to include a prospective use. In contrast, a cost overrun is an ex-post event and affects only one of the original set of potential projects, namely the set actually constructed. Some methods (particularly those relating to ex-ante planning techniques) require that the costs be estimated for every possible hypothetical project involving each single use and all combinations of uses up to and including the "grand project" incorporating all uses. Other methods (especially ex-post) require only a subset of this information. The general idea for ex-ante planning is to calculate and subsequently combine information on how much it costs to serve each purpose singly and to add each purpose to some combination of the other j purposes, producing a j + 1 purpose project. In particular, the cost of adding each purpose to the N-l other purposes, and thus arriving at the grand N-purpose project, will always be calculated.

The Core, a Concept from Game Theory

The concept of the core is very closely linked to game theory. In the jargon of the field, the question is: does the allocation reside in the core of the game defined by the (endogenous) cost information on possible alternative, stand-alone projects serving the same purpose? In more intuitive terms, this characteristic involves asking whether the representative of every purpose would have an incentive to participate in the project. The requirements for this to be true can be stated verbally as follows.

(a) Does the allocation just exhaust total costs?

(b) Does it allocate to each purpose or group of purposes no greater cost than that for which the purpose or group could be served by a project built to serve it alone?

(c) For any J-purpose project, does it allocate to each purpose or group of purposes no less than the marginal cost of adding the purpose or group to a project built to serve the subset of purposes excluding it?

Formally, call a single purpose i, a subset of purposes S and the full set of possible purposes N. Let C(i) equal the cost of a single purpose project, C(S) the cost of a project serving S, and C(N) the cost of the grand project. Let X_i be the cost allocation to purpose i. Then (a), (b) and (c) can be written:

(a) $\sum_N X_i = C(N)$ (A-1)

(b) $\sum_S X_i \leq C(S)$ for all S in N including single purposes (A-2)

(c) $\sum_S X_i \geq C(N) - C(N - S)$ for all in N including single purposes (A-3)

In fact, (b) and (c) are equivalent when (a) holds.

An allocation that meets these requirements gives no incentive for a single purpose or group of purposes to "go it alone" by building a separate project. Neither does it create a grievance by subsidizing one purpose or group at the expense of others. It would therefore be desirable always to be in the core for reasons of equity and incentive. but it should be noted that:

- the core may not always exist; no allocation may be possible that fulfills conditions (a) - (c);
- even if the core is non-empty, not every method produces allocations that are in it;
- when the core is non-empty, it will in general contain infinitely many points, so that further choice is necessary.

Some other commonly mentioned criteria on which cost allocation methods may be judged include the following.

Additivity, which covers the possibility of adding up sub-allocations. For example, if capital and operating costs are distinguished and allocated separately, will the total resulting allocation be the same as the one that would result from an allocation of the total of those costs?

Monotonicity, which covers the effect on individual cost allocations of increasing *ex-post* either the total costs of the grand project or the actual costs of individual purposes or groups. In the present context, the most significant version of this requirement is that if the costs of the grand project are greater *ex-post* than estimated for the allocation exercise, no purpose or group of purposes will have a lower allocation as a result of having this taken into account.

Consistency, which deals with the effect of "removing" a purpose by making the implied allocation to it and then reconsidering the allocation for the N-l remaining purposes. The method is consistent if the reconsideration leads to the same allocation for the N-l as did the initial application.

Covariance, which requires that a method give strategically equivalent allocations (allocations with the same incentives for behavior in the cost game) in whatever units the costs are measured and from whatever baseline they are measured.

It is important to note that, when taken as a group, these last four criteria are seldom met even by complex, infrequently applied, *ex-ante* game theoretic methods. If one reflects a moment on what the criteria mean, it is readily apparent why this should be the case. For all four to apply, one would need to devise an allocation method that did all of the following.

1. Costs could be differentiated in any arbitrary fashion and still result in the same allocation (additive).

2. Any arbitrary cost over-run could be incurred without reducing any group's cost allocation (monotonic)
3. Any purpose could be removed and not affect any of the remaining purposes (consistent).
4. Costs could be measured in any units, from any baseline, and still result in the same strategy for all groups of participants (covariant).

To the best of our knowledge, no method of cost allocation meets all these criteria, and so one must choose a method that is appropriate for a particular problem.

Categories of Joint Costs

Another observation is that the costs of increasing fish runs reflect facility investments and changes in operating procedures that stand in a variety of relationships to the original purposes of the system and its component dams.

1) Some are specific to a dam and a purpose. Examples include screens on turbine penstock intakes, which serve only the hydropower function of the dam where they are installed.
2) Some are specific to a dam but joint for the multiple original purpose of the dam. For example, fish ladders and redesigned spillways, while serving only one dam, are needed to reduce losses due to most of the dam's original purposes, including hydropower, navigation, and irrigation.
3) Some are joint for the system of dams and the multiple uses of the system. Changes in operating rules designed to increase flows during periods of downstream migration of smolts (to reduce reservoir mortality) fall into this category.

For purposes of this appendix it is assumed that costs of the first type will be allocated to the appropriate account and that the allocation will not require detailed analysis. It is further assumed that the proposed system simulation model described in Volume 2, Part II will allow the allocation of the third, or system-wide, cost to dams. [Note that many policy choices need to be made before this can be done.] Therefore, the principal allocation question considered here is:

> How can BPA allocate an ex-post overrun in the cost of each system dam to the multiple uses of each dam?

The rest of this appendix describes alternative answers to this question.

APPLICABILITY OF ALLOCATION METHODS TO FISH MITIGATION

In order to choose a method that best applies to the fish mitigation problem, one must first try to characterize the problem accurately in the terms of the characteristics outlined above, taking institutional and legal constraints into consideration. In this case, it is convenient to take legal constraints first.

Legal Constraints

As already noted, the Regional Act requires that costs be allocated based on existing procedures. The procedures used when the various projects were put in place was Separable Cost Remaining Benefits (SCRB). This method, explained below, while generally used in water resources planning has recently been criticized as a tool for ex-ante cost allocation, the most common problem considered by cost allocation specialists (H. P. Young, N. Okada, and T. Hashimoto, "Cost Allocation in Water Resources Development," Water Resources Research, vol. 18, pp. 463-475, June 1982). However, since its use is specified in the legislation, the next step would seem to be to decide whether or not it has the characteristics described above in the present context, keeping in mind that this is primarily an ex-post accounting exercise and not a "strategic" situation. The next section develops the verbal and mathematical basis of SCRB, and sets out some notation used in assessing the characteristics of SCRB in terms of additivity, monotonicity, and the other criteria.

SCRB

This section will lay out a verbal version of the SCRB method and then display the commonly accepted formula by which the words are translated into a working method. Then it will show how a cost overrun enters, and the formula for ex-post cost accounting.

The verbal version is taken from "Economic and Environmental Principles and Guidelines for Water and Related Land Resources Implementation Studies", March 10, 1983, p. 14.

(a) Separable cost for each purpose in a plan is the reduction in financial cost that would result if that purpose were excluded from the plan. This reduction in cost includes:

(1) The financial cost of measures serving only the excluded purpose; and

(2) Reductions in the financial cost of measures serving multiple purposes. In some cases removal of a purpose would result in selection of different measures to address the remaining purposes.

(b) Joint cost is the total financial cost for a plan minus the sum of separable financial costs for all purposes.

(c) Alternative cost for each purpose is the financial cost of achieving the same or equivalent benefits with a single-purpose plan.

(d) Remaining benefit for each purpose is the amount, if any, by which the national economic development (NED) benefit or, when appropriate, the alternative financial cost exceeds the separable financial cost for that purpose. [NED can be defined as the total consumer surplus that will result from a project]. The use of alternative cost is appropriate when alternative financial cost for the purpose is less than the NED benefit, or when there are project purposes that do not address the NED objective.

Costs allocated to each purpose are the sum of the separable cost for the purpose and a share of joint cost as specified below:

(a) Joint cost may be allocated among purposes in proportion to remaining benefits.
(b) Joint cost may be allocated in proportion to the use of facilities, provided that the sum of allocated joint cost and separable cost for any purpose does not exceed the lesser of the benefit or the alternative cost for that purpose. [Exogenous method]

Formulae. In translating these verbal descriptions into formulae, nothing is lost by assuming that the benefits of every purpose are greater than its alternative costs (the costs of a stand-alone project). That is, it is assumed below that the SCRB cost allocations performed when the various Columbia River projects were constructed were done "correctly", and that the alternative costs for each purpose and project exceeded their separable costs. This assumption greatly simplifies the development of the equations, although the final results do not depend upon the assumption.

Let N be the number purposes potentially to be included; index N by i. Indicate the cost of a project by $C(\cdot)$, where the argument can indicate a single purpose project, C(i), a project serving $S \leq N$ purposes C(S), or a project serving all but one of S ($\leq$ N) projects C(S − i).

Then, the separable cost for purpose i may be written SC(i) = C(N) - C(N-i). The alternative cost of i is written C(i) for purpose i. Joint or nonseparable costs are defined to be NSC = C(N) - $\Sigma SC(j)$.

We can write the formula for the total cost allocation to purpose i as:

$$X(i) = C(N) - C(N-i) + \frac{C(i) + C(N-i) - C(N)}{\sum_j (C(j)+C(N-j)-C(N))} \left[\sum_j C(N-j) - (N-1)C(N)\right]$$

Now, assume that the actual cost of the full project is not C(N) but C(N) + F then it is straightforward to show that the new cost allocation X'_i is given by:

$$X'(i)=C(N) - C(N-i) + \frac{C(i) + C(N-i) - C(N)}{\sum_j (C(j)+C(N-j)-C(N))} [\sum_j C(N-j) -(N-1)C(N)+F]$$

Next, for notational convenience, let

$$a = C(N) - C(N-i)$$

$$b = \frac{C(i) + C(N-i) - C(N)}{\sum_j (C(j)+C(N-j)-C(N))}$$

and

$$d = (\sum_j C(N-j) - (N-1)(C(N))$$

Then it follows that

$$X(i) = a + (b * d)$$

and

$$X(i)' = a + (b * (d + F)) \qquad \text{(A-4)}$$

This notation will be used below to demonstrate some of the properties of SCRB.

<u>The Core and SCRB</u>

First, consider whether or not SCRB will result in a solution that is in the core, in the case where an equal cost overrun, F, occurs for all sub-projects, the original allocation is in an existing core, and the core exists after accounting for the <u>ex-post</u> cost overrun. Again, let C(i) equal the cost of a single purpose project, C(S) the cost of a project serving S, and C(N) the cost of the grand project. Let X_i be the cost allocation to purpose i. Modifying equations (A-l) through (A-3), to account for the cost overrun (i.e., the mitigation costs) gives :

$$\sum_N X_i + F = C(N) + F \qquad (A\text{-}5)$$

$$\sum_S x_i + F \leq C(S) + F \text{ for all } S \text{ in } N \qquad (A\text{-}6)$$

$$\sum_S X_i + F \geq [C(N)+ F] - [C(N - S)+ F] \text{ for all in } N \qquad (A\text{-}7)$$

Thus, assuming an equal cost overrun for any project, the method does give results that are in the core, assuming it exists.

Additivity

SCRB applied to an ex-post overrun will, in general, generate allocations that exhibit additivity. To see this, consider the equation for X(i)', (A-4):

$$X'(i) = a + (b * (d + F)).$$

Now, assume that F is divided into i separate (and not necessarily equal) categories, such that $\Sigma F_i = F$. Then it is easy to see that if

$$X''(i) = a + (b * (d + \Sigma F_i))$$

Then

$$X'(i) = X''(i).$$

Thus, the SCRB/overrun allocations are additive.

Monotonicitv

In order for the SCRB/overrun allocation to be monotonic, no purpose or group of purposes may have a lower allocation as a result of the overrun. Since, in equation (A-4) all of the terms are positive, X'(i) will always be positive. Therefore, distribution of the cost overrun is monotonic when the goal of the allocation exercise is ex-post accounting for the cost overrun, which is the goal that seems reasonable in the Bonneville context.

Consistency

Consistency refers to the effect of removing a purpose from the project, then re-allocating the costs to the remaining projects. This does not apply in an ex-post context, since the "original" purposes cannot, as a practical matter, be removed from the (existing) project.

Covariance

The effects of changing the base period, the units in which costs are measured, and so forth are, in an ex-post context, policy decisions outside the cost "game". As such, game theory has no application here.

Summary

Based on the characterization of the problem given above and on the legal requirements in the Regional Act, it appears that the SCRB method required by law is compatible with applicable game theoretic concepts. Therefore, our preliminary conclusion is that there is no reason not to use SCRB to allocate mitigation costs. Although it is far from perfect in many applications, it is a reasonable way to structure the inherently ambiguous process of cost allocation.

CONCLUSION

Research in Phase II has permitted an examination of the literature on joint cost allocation and some extensions of it to the Bonneville context. But examination of the literature and reasoning about it, because of the nature of the problem, cannot yield an unambiguous result. The approach in Phase III will be to use the proposed simulation model described in chapter 2 of Part II to experiment with a variety of methods to try to build a consensus about what is a reasonable system to implement. Upon an initial look procedures incorporating the SCRB concept appear to be desirable legal grounds. The preliminary examination of SCRB methods, based on game theoretic concepts, concludes that they are adequate for the task, when combined with a detailed simulation model.

PART III

OCEAN FISHERIES HARVEST MANAGEMNET

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Part III

Ocean Fisheries Harvest Management

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PREFACE TO PART III

This Phase II report deals with the modeling and economics of management of ocean fisheries based on Columbia River Salmon.* The research it proposes promises to yield a useful general set of tools for the analysis of ocean fisheries management alternatives. A chief purpose is to supply estimates of the opportunity costs (net benefits foregone), of different levels of regulation of ocean fisheries, that can be used to make comparisons with the costs of other mitigation alternatives, aimed at increasing upstream runs, in a cost-effectiveness framework. No benefit analysis of the upstream runs is implied by this approach and none is contemplated in the Phase III research. The research approach reported here is responsive to task one in the work plan for Phase II.

Task I calls for investigation of the feasibility of, and development of a plan for, a system model including the hydrologic, ecologic, and economic components of the Columbia River system. Where suitable, components of existing models are to be included. The proposed model, or set of models, was also to incorporate alternative mitigation strategies, including _harvest management strategies_, for the purpose of comparing the cost effectiveness of such alternatives.

This volume is divided into four main substantive sections:

1: Historical overview of the Columbia River ocean fisheries;

2. The conceptual delineation of the recommended research approach;

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* This volume is adapted from a report submitted to Resources for the Future by Virgil Norton, currently at the University of West Virginia. Ian Hardy contributed to chapter 111-2, Nancy Bockstael, Kenneth McConnell and Ivar Strand to chapter 111-3, Mae White to typing the report, and Nancy Anders Norton to the entire report.

3. A review of relevant literature and specification of the methodology for estimating net economic values of recreational and commercial ocean fisheries; and
4. A summary of the concept of economic impact and the appropriate approach to determining economic impacts.

An addendum is included that proposes an approach to Phase III research in the ocean fisheries area. A second addendum lists persons consulted in the preparation of this volume. The volume closes with a list of references.

Chapter 1

History of the Columbia River Salmon Fisheries

In order to understand the current fisheries based on Columbia River salmon stocks, it is important to have a general historical perspective of how the fisheries evolved over time. We likely tend to think of the commercial and recreational exploitation of Columbia River stocks as a relatively recent -- i.e., 19th and 20th century -- development. However, native Americans fished for both commercial and sport purposes and utilized the salmon stocks of the Columbia River hundreds of years ago. (Much of the information in this section is from Netboy, Smith, Crutchfield and Pontecorvo; and Northwest Power Planning Council).

Explorers Lewis and Clark and David Thompson, in reporting upon their experiences, related considerable information concerning the harvest and use of the salmon stocks by native Americans. It has been estimated that throughout the Snake and Columbia Rivers and their various tributaries, there were more than 50 thousand native Americans. It is further estimated that these individuals utilized an average of one to two pounds of salmon per day per person. This is the equivalent of 18 to 36 million pounds a year. The native Americans utilized salmon for food, trading, fuel, dog food, and cultural and religious ceremonies. There are many legends about the salmon and other wildlife such as the coyotes and their role in the culture, religion and survival of the native Americans. The catch of this relatively large amount of salmon was spread throughout the habitat and apparently was compatible with the biological capabilities of the species.

During the primary up-river migration season for adult salmon, the human population along the Columbia River would increase, perhaps tripling. Native Americans from the Great Plains and the Rocky Mountains came to the Columbia and Snake River areas to trade deer and buffalo hides for dried salmon. The dried salmon were of ten packed in bales of 90 to 100 pounds. It is reported that this process was well organized and that the local Columbia River and Snake River inhabitants would sell fishing rights and the use of their fishing gear to the visitors. This time of the year was an important social, commercial and religious event for these native American peoples.

In the early 1800s, the Hudson Bay Company began to expand operations in the Columbia River area. Along with this expansion came the first of a series of developments that led to a diminishing of the use of the Columbia River stocks by the native Americans, and an expansion of operations by the European immigrants. By 1829, businessmen with names such as Hume and Wyeth from New England were involved in the Columbia River salmon fisheries in a major way. At that time, moderate amounts of salted salmon were being shipped to California; Valparaiso, Chile; London; and Honolulu. Salmon trade was becoming an important commercial venture -- commercial in the terms that we generally use it today.

At this point, with the influx of new immigrants into the Pacific Northwest, the native Americans were undergoing a substantial population decline which was caused by diseases and other circumstances. During this period the Columbia River salmon populations apparently increased. This was because of the substantial decline in the native American populations and the still relatively small use of the salmon stocks for “modern” commercial ventures. In the mid 1800s, the salmon populations were perhaps at the highest level for centuries.

Around 1865, however, even more significant changes began to occur. At this time salmon canning was introduced and almost overnight Columbia River salmon products were sold in worldwide trade. These were products ranging from luxury high priced items to low cost food for factory workers in England. At this point, in response to the apparent market potential, new fishing technology was introduced. This involved gillnets, seins, fishwheels, and fishtraps. As a result, catches of salmon increased substantially. At the same time, there was considerable cultural stratification taking place in the area.

White cannery workers were considered by cannery owners as unreliable and transient. However, Chinese workers were highly valued and relatively well paid. This led to a situation of strict ethnic lines for activities related to the exploitation of Columbia River salmon. The cannery workers were oriental and the fishermen were occidental. There was strict enforcement of these ethnic lines. It is estimated that by 1880 there were 2,500 fishermen (non-native Americans) exploiting the salmon on the Columbia River. They were a special breed. Most of them were unmarried and lived in boarding houses. They were independent and a tough group of individuals, and there were many conflicts regarding river resources.

Estimates are that by 1900, there were 2,800 gillnetters operating on the river. The Columbia River Fishermen's Protective Association was formed to advance the interest of the gillnetters. The membership of this union was restricted; the meetings were lengthy and closed to non-members. Membership requirements for this association included the exclusion of liquor dealers, capitalists, lawyers and politicians. Union strikes were frequent; and snag vessels were used to eliminate the equipment of nonparticipants and of fishermen other than gillnetters. The gillnetters made many efforts to pass legislation to eliminate other forms of fishing. To this point however, they had not been successful.

In 1902, the stage was set for further changes. Voters in Oregon passed a constitutional amendment to provide for statewide initiatives and referendums. As a result, the people of the State of Oregon could actually both make and veto laws. Through this process the gillnetters, with backing from the Grange, the Oregon Federation of Labor, and the Oregon Fish Commission, placed laws on the books designed to eliminate fish-wheels, fish-traps and forms of fishing other than gillnets. This was done even though fish-wheels and fish-traps are an efficient method of taking salmon.

The important point here is that the gillnetters were able to win this battle for the fish in the Columhia River primarily because, in numbers, they were the largest group. This, unfortunately, is just the opposite of what might have been most desirable from the standpoint of efficient use of a common property resource. That is, a large number of fishermen, using an inefficient harvesting technique were now the primary harvesters of the Columbia River salmon resource.

As more and more restrictive legislation and regulations were passed relative to the use of the salmon stocks on the River, an important technology development occurred. The gasoline engine was developed to the point where it could be adapted for fishing vessels. This started a substantial move out beyond the mouth of the Columbia River into the ocean for salmon fishing. A primary purpose was to escape from the restrictive regulations regarding gear and other activities on the River. The off-shore fishing technique (trolling) is also inefficient when compared to possible river methods such as fish-wheels or traps.

Therefore, the Columbia River salmon commercial operations in the river (gillnetters) and off-shore (trolling) evolved to the present day situation of a large number of operators using inefficient fishing methods. Since the early 1900s to the present time, the Columbia River based fisheries have been affected by expanding fishing effort and the imposition of a multitude of complex fishing regulations. This situation along with various other factors (logging, dams, pollution, fishing by foreign fleets, and other possible causes of declining populations of salmon) were the impetus for the development of a number of attempts to improve the salmon management scheme.

These factors culminated, in 1976, in the passage of The Fishery Management and Conservation Act. This Act was passed for the purpose of attempting to rationalize domestic fishing effort and (probably primarily) to control foreign fishing within two hundred miles of the U.S. coasts. Regional Fisheries Management Councils were established to carry out the requirements of the Act. Included among the eight councils was the Pacific Regional Fishery Management Council which has the responsibility for managing, among other species, the salmon stocks that are exploited off the coasts of Oregon, Washington, and California. There has been considerable controversy regarding the Councils and their effectiveness. Certainly, the management of salmon under the Council is complex and difficult. One reason is that the Council has no genuine authority within the three-mile territorial sea. These areas are under the authority of the state management agencies. Therefore, considerable coordination, cooperation and compromising has been necessary in order to develop the Fishery Management Plans that now control the harvesting of salmon stock off the coasts of three states.

The management of Columbia River salmon is also complicated because of the ocean migratory pattern of the stocks. The Chinook, for example, migrate as far north as southern Alaska. This means that the Columbia River Chinook and Coho are intermixing with other salmon populations and species up and down the coast. The harvest in a particular area, therefore, will generally include individual fish from a number of different populations. Interaction among these populations in a given fishing area represents a complication in the management process. That is, if one stock is relatively weak while another stock is strong, normal fishing on the strong stock may tend to over-fish or even eliminate the weak stock. The same is true, of course, with respect to wild and hatchery stocks. When many hatchery fish are stocked in the river, they intermix with less numerous wild stocks. The allowable harvest on the hatchery stocks can result in serious over-fishing and further declines in the wild stocks. In practical terms, what this means is that heavy fishing off the coasts of British Columbia and Southern Alaska has impacted on the Columbia River stocks, especially Chinook. It has been difficult for fishery managers to attain an effective agreement to reduce fishing on the Columbia River stocks in the northern areas because reducing the allowable harvests of Columbia Chinook stocks meant a substantial reduction in the harvest of Alaskan and Canadian stocks. This was especially complex because the agreement of both the Canadian fishery management agency and the agencies involved in the Alaskan fisheries was essential. Each would not agree, however, without assurances that if fishermen in their area were restricted, fishermen in the other areas would likewise be restricted. This is important because half or more of fish originating in the Columbia River are caught off the coasts of Canada and Alaska.

However, in the mid 1980s a treaty was signed between Canada and the United States that may make effective regulation of the Pacific salmon fisheries possible. The previous lack of an enforceable agreement relative to the Canadian and Alaskan catch of Columbia River stock made any rational mitigation or enhancement plan impossible. Now, however, with the treaty, and the possibility of establishing appropriate harvest restrictions off the coasts of Canada, Alaska, California, Oregon, and Washington, it should be possible to develop regulations that will enhance upstream runs as salmon stocks are increased. This possibility makes improved understanding of the economics of the ocean fishery especially important.

CURRENT COLUMBIA RIVER SALMON FISHERIES

The commercial salmon fishery based on Columbia River stocks is made up primarily of Chinook and Coho. In examining the role of the ocean fisheries, it is important to identify the role of salmon from the Columbia River relative to the overall supply of fish products. It should be noted that in the United States approximately 50 percent of all fish products are imported. United States salmon landings make up about 16 percent of the value of U.S. fish and shellfish landings and approximately one third of the value of the U.S. finfish catch. Relative to total fish consumption, the importance of U.S. caught salmon would be less because of imports. The Columbia River contributes about 6 percent of the value of all salmon, approximately 0.7 percent of the value of all fish and shell fish landings in the U.S. and, therefore, about 0.3 percent of U.S. fish consumption. Thus, the catch of salmon from the Columbia River, while extremely important locally, does not contribute in a significant way to the overall U.S. fish consumption. In terms of importance by species, Columbia River Chinook make up nearly 40 percent of all Chinook caught and Columbia River Coho make up about 20 percent of all Coho caught.

One element of particular importance relative to the commercial salmon is the balance of trade. The exports of salmon products are nearly nine times that of the imports of salmon products. In fact, the value of salmon products from throughout the U.S. makes up more than 50 percent of the value of the exports of all U.S. edible fish products. Exported Columbia River commercially-caught salmon products are, however, a relatively small percent of the total salmon exports.

There are more than 5,000 trollers operating off the coasts of Oregon, Washington, and California. This calculates out to a per-vessel average value of salmon catch of below $5,000. Ninety percent of the salmon troll catch is made by about 2,000 vessels, representing an average value of catch of approximately $10,000 per-vessel. Even more revealing, is that only 500 vessels (approximately one tenth of the total) harvest 50 percent of the fish and that this represents an average annual value of catch per vessel of only about $20,000. Therefore, it is clear that most of the vessel owners operating in the coastal waters of Oregon, California, and Washington are part-time fishermen at least in these waters; some fishing only a day or two a year. Many of these individuals have full-time jobs elsewhere and others are primarily fishing in other waters, such as off of Alaska.

The recreational activities off the coast of Oregon, Washington, and California are substantial. It is reported that there are more than one-half million angler trips per year in the ocean areas. These are made up of private skiffs or pleasure boats, and charter vessels. In California, the division is nearly equal between the charter vessels and private boats. In Oregon, however, most of the angler trips are on non charter vessels. In Washington, fisher-men trips on charter vessels out number those on pleasure boats.

In terms of number of fish caught, the commercial catch is substantially higher than the recreational catch off the coast of California and Washington, and in the Columbia River. This is also the case for- Chinook off the coast of Or-egon. However, the recreational fishery takes the largest number of Coho off the Oregon coast . Overall, the commercial catch accounts for almost two-thirds of the total number- of Coho and Chinook caught commercially and recreationally in the ocean and Columbia River- fisheries.

As was indicated earlier, the effort in both the recreational and commercial fisheries has increased over the past several decades. This has resulted in greater and greater restrictions relative to commercial and recreational harvesting. This is particularly evident in terms of the number of days of fishing allowed for salmon off the coasts of Oregon, Washington, and California. For example, in the late 1970's the allowable recreation fishing days were approximately 185, with bag limits of up to three salmon. In recent years, allowable days have been lowered to about 40, bag limits have been reduced and additional restrictions have been imposed. The reduction on the commercial side is even greater. In the commercial ocean fisheries in the late 1970s, the number of allowable fishing days for at least one salmon species was nearly 140. In recent years this has been reduced to less than 10. The decline in legal fishing days in the river commercial fishery has been relatively less, from approximately 40 to 20. These reductions in allowable fishing days and allowable catch are a reflection of the fact that the existing commercial and recreational effort is substantially greater than would be required to take the allowable catches with fishing seasons the length they were a decade ago. This is a typical result with a common property resource when entry into the industry is not restricted.

Chapter 2

Mathematical Programming Models of the Ocean Salmon Fishery

INTRODUCTION

Mathematical programming offers a potentially attractive way to model the bio-economics of the ocean fishery for salmon. It also provides a means to explore the effect of different management alternatives on these relationships. Formally, the mathematical programming model is a constrained-optimization problem that is stated in terms of single-valued continuous differentiable functions. This type of model offers limited opportunities to consider stochastic elements of the fishery problem. However, it has the potential capability of handling large variable set (Hilgaer, et al.) and possesses other attributes which enhance its desirability as a way to quantitatively represent the ocean salmon fishery.

In the following, we will discuss how mathematical programming might be applied to the salmon ocean fishery problem. We shall also assess the pros and cons of some alternative formulations for this class of models, and propose a particular formulation which we believe strikes a reasonable balance between ease of computation, flexibility of use and richness of results. The assumptions necessary to develop this type of model are specified, and its desirability as a way to model the fishery is evaluated.

The mathematical programming approach has four key advantages. It offers a practical way to bypass the parsimonious use of variables required by other approaches. It is oriented towards exploration of the effect of different management policies on key variables rather than towards the estimation of parameters. It allows explicit formulation of the criterion to be used when choosing between feasible alternatives. It also provides a relatively rich set of results, including consistent and interpretable shadow values to the ocean fishery for the salmon entering the ocean fishery and those returning to the river. Such shadow prices would be essential in evaluating policies aimed at increasing or maintaining salmon stocks.

Policies developed to change recruitment or escapement are likely to require a detailed representation of the types of salmon present in the ocean fishery. Such policies are likely to focus on enhancement of wild populations or on increased releases of hatchery stock. Thus, the spawning source needs to be distinguished in a salmon fishery model. Management programs also will be concerned about population differences among rivers. Hence, spawning location and species composition become important. The age or size of salmon also are important: policies that increase fishing in the lower Columbia River will, for example, increase the catch of older larger salmon relative to policies which encourage off-shore fishing. Given such policy options, salmon would need to be distinguished by spawning source, spawning location, species, and age or size. It would also be necessary to separate the successive runs or cohorts of salmon entering the ocean fishery over time.

Faced with so many attributes, homogeneous variable definitions can be obtained only by defining large variable sets. Suppose, for example, salmon sub-populations are defined for six ages, two species, two spawning periods, five spawning locations, and twelve successive runs or cohorts. 1,440 variables would then be required to completely describe the salmon fishery's population. Mathematical programming models have the potential to consistently evaluate such sets of variables. Most other approaches would require either aggregation into more heterogeneous groupings or the definition of complicated variables.

When a large number of variables are defined, simulation and mathematical programming gain prominence in the list of possible modelling approaches. Simulation offers a positive ad hoc approach which can include stochastic elements. Identification of good or better solutions is often a matter of judgement, however, and the reason why a particular- solution has emerged is often difficult to extract from the "black boxes" of a large simulation model. An extensive discussion of the relative strengths and weaknesses of simulation models is found in Volume II. Mathematical programming is a normative deterministic approach in which an explicit objective function is specified, and solutions are unambiguously ranked according to a particular value. The explicit objective function provides the necessary foundation for the computation and interpretation of shadow prices for the uncaught salmon. As noted before, these values are a key benefit of the mathematical programming alternative.

Although shadow prices are one of the most valuable results from a programming model, they are only one of several potential outputs. A mathematical programming model of the ocean salmon fishery would determine whether a set of escapement goals could be feasibly attained when recruitment and catch are at particular levels. It would provide a profile of an optimum harvest , and could show how this harvest would be affected by various controls on fishing effort or intensity. The model could also be used to isolate the effect of changes in recruitment levels on the number and type of salmon caught, the value of the fishery, and the number of salmon escaping to the river. We believe this set of potential results is rich enough to justify consideration of a mathematical programming formulation of the ocean fishery.

GENERAL MODEL STRUCTURE AND APPROACH

The heart of most bio-economic analyses is a fish population dynamics system which relates harvest or catch to fish stocks (Clark). Stocks are, as Beverton and Holt pointed out many years ago, dependent on recruitment, growth , natural mortality, and fishing mortality. If fishing mortality is replaced by effort and intensity (where effort measures the catching power of an individual vessel and intensity measures the number and composition of vessels in a given area at a given time), catch may be r-elated to fish stocks and harvest effort in a single relationship. Chambers and St t-and, for example, express the harvest level (h_t) as

$$h_t = \frac{c \cdot E_t}{c \cdot E_t + M} (1 - \tau_t) X_t \tag{2-1}$$

where X_t is the size of the resource stock at time t, E_t is a vector of effort levels from different gear, c is a vector of instantaneous rates of mortality due to a unit of effort, M is instantaneous natural mortality, and τ_t is the proportion of current population transmitted from period t to period t+l.

Most programming models would separate the effect of fishing effort and intensity from the effect of fishing stocks on harvest. Fishing effort and intensity would appear in the objective function in the form of a cost or supply function. The effect of stocks on catch would be represented in the constraint set of the model. This separation follows the traditional economic approach of maximizing or minimizing an economic value subject to a given input-output relationship. Objectives that would fit this particular model structure would include minimization of the costs of harvest, maximization of the value of the catch and maximization of the economic surplus created by the fishery. A similar structure could serve for a model that maximizes catch or maximizes escapement for a given catch. The separation of harvest effort and intensity from the population dynamics is a distinguishing feature of a mathematical programming model for a fishery (Li and Williams; Rothschild and Balsigner; Siegal, et al.).

Management controls or programs can be incorporated into the mathematical programming model either as part of the constraints or as part of the formulation of the objective function. Constraints establishing minimum escapement goals or recruitment levels are examples of the first alternative. Development of harvest costs or demand functions based on a given division of catch between commercial and sport fishermen would be an example of the second. Some management policies must be analyzed by changing one or more parameter values and either re-solving the problem or performing a post-optimality analysis. Change in specified recruitment levels would be one example of this third alternative. Between the three alternatives, most management policies can be introduced into and analyzed through a mathematical programming model.

In the following, we first define a set of variables that can reasonably describe the population of salmon present in the ocean fishery. We then introduce a simple numbers-of-fish accounting system and develop it into a set of linear constraints describing the fishery's population dynamics.

DESCRIBING THE SALMON POPULATION

Perhaps the first task in developing a mathematical programming model is to decide how to describe the salmon population. As noted in the introduction, one may either define homogeneous subpopulations or develop a set of complex variables with grouped or heterogeneous attributes. Since a primary advantage of mathematical programming is its capability of handling large variable sets, we focus on the definition of homogeneous subpopulations.

In the following discussion, the letter x will be used to denote a quantity of salmon caught from the ocean fishery. The letter y will denote a quantity of salmon that remains uncaught. Total population can be represented by summing x and y. Subpopulations can be defined by appending subscripts to these variables. Three subscripts will be distinguished: the type of salmon to which the variable applies, the age of the salmon to which the variable refers, and the time period in which the variable is relevant. Thus, x_{iat} will represent the quantity of salmon of type i and age a caught in time period t, and y_{iat} will represent the quantity of salmon of type i and age a which is not subject to fishing mortality in period t.

The subscript t will be interpreted as a discrete variable enumerating a set of dates spread along a time continuum. The time between the two successive dates t and t+l will be referred to as time period I,. The time "interval" will be used to designate two or more successive periods. We shall assume the dates are spread widely enough along the continuum to allow the assumption that all salmon from a particular run enter the ocean fishery on a particular date. For convenience, the start of a planning period or other key interval will always be designated by t=1.

The term “type of salmon” is used to refer to any homogeneous subgrouping of fish that is distinguished by species, spawning location and spawning source. Other attributes of the salmon in the fishery may be used to distinguish between types if necessary. For example, one might distinguish between salmon caught in different geographic areas within the fishery. The set notation i is contained in I will be used to refer to the set of all possible types of salmon.

Although the subscript variable "a" technically may take any non-negative integer value, it will be convenient to limit this variable to the integers 1 to 6. This implies that all salmon return to the river to spawn before age 7. Age will be set to 1 when the salmon arrive in the ocean fishery, and it will be assumed that some salmon return to the river at ages 3, 4, and 5. Thus, tenure within the ocean fishery will vary from 3 to 6 years. This tenure designation may be lengthened or shortened in the model, depending on the type of salmon considered.

For convenience, we shall refer to the salmon recruited at date t as “cohort” t. This designation will be maintained as long as the fish remain in the fishery.

MODELING THE POPULATION DYNAMICS OF THE FISHERY

A key task in developing a mathematical programming model is the construction of a set of constraints describing the population dynamics of the ocean fishery. Since the number of homogeneous subpopulations is expected to be large, the method to be used in solving the programming model becomes an important consideration when formulating the population dynamics system. If the constraint set is linear in both variables and parameters, a linear or quadratic model can be used. A linear programming model would place minimal restrictions on the number of defined subpopulations, since such models may contain thousands of variables and still be solved by the Simplex Method. This will not be true for quadratic programming, although the possibility exists that this type of model may be solved for enough salmon subpopulations to make it a useful alternative. Other mathematical programming models are less promising, since they tend to be restricted in size by available solution algorithms. The number of feasible homogeneous salmon subpopulations would be too small to successfully describe the ocean salmon fishery .

Given these considerations, it seems reasonable to model the population dynamics of the salmon in a way that is consistent with the use of linear and quadratic programming. Thus, we formulate a number of fish equation systems based on a set of discrete time periods. This system can be expressed in a linear form, or it can be generalized to allow for natural survival rates which are dependent on population density. The latter option is only feasible, however, if the equation system is extended into a nonlinear form. Such a nonlinear extension is outlined later in the paper. Initially, we discuss the linear form which is required for the linear and quadratic programming options.

The linear system is developed in two steps. First, a given cohort is consider-ed. Then a set of constraints is developed for all cohorts in a given time interval. Type of salmon is initially restricted to a single type in both cases. More than one type of salmon is considered in a later section where programming models based on the linear constraint set are presented.

POPULATION DYNAMICS FOR A GIVEN COHORT

The population dynamics for a given cohort and type of salmon may be expressed for any particular- set of time periods, provided these periods span the interval that the cohort resides in the fishery. For our purposes, a six-year interval is used because we assume some salmon remain in the ocean fishery until age 7 before returning to the river. (As noted earlier, the assumption regarding age may be relaxed, if needed.) Also, when several cohorts enter the fishery within a year, t can be defined for enough periods that each cohort may be assumed to be recruited on a date represented by a t. These restrictions create the basic structure of the population dynamics system for a given cohort and type of salmon.

Given this basic structure, we may formulate a simple accounting system relating the number of salmon caught during period t to the number of uncaught salmon that survive from period t to period t+l. Keep in mind that while it is assumed here that each period represents one year, this could be altered so that t can represent any period length. Let $t=t_1$ be the beginning date of the first year, $t=t_2$ the beginning date of the second year, $t=t_3$ the beginning date of the third year and so on the for the six years that the cohort resides in the ocean fishery.[1] Then the accounting system may be expressed as :

$$x_{i1t} + y_{i1t} = \mu_{it} \qquad t = t_1 \qquad (2\text{-}2)$$

$$\alpha_{i1t} y_{i1t} + \beta_{it} x_{i1t} = x_{i1\,t,+1} + y_{i1\,,t+1} \qquad t = t_1, \ldots, t_2 - 2 \qquad (2\text{-}3)$$

$$\alpha_{i1t} y_{i1t} + \beta_{it} x_{i1t} = x_{i2,t+1} + y_{i2,t+1} \qquad t = t_2 - 1 \qquad (2\text{-}4)$$

$$\alpha_{iat}(1 - \tau_{iat}) y_{iat} = x_{ia\,t+1} + y_{ia\,t,+1} \qquad a = 2, \ldots, 5 \quad t = t_a, \ldots, t_{a+1} - 2 \qquad (2\text{-}5)$$

$$\alpha_{iat}(1 - \tau_{iat}) y_{iat} = x_{i,a+1,t+1} + y_{i,a+1,t+1} \qquad a = 2, \ldots, 5 \quad t = t_a - 1 \qquad (2\text{-}6)$$

$$Y_{iat} \geq 0 \quad x_{iat} \geq 0 \qquad \forall\ i,a,t \qquad (2\text{-}7)$$

$$\tau_{i1t} = 0 \quad \tau_{i6t} = 1 \qquad 0 \leq \tau_{iat} \leq 1 \text{ when } \quad a = 2, \ldots, 5\ \forall\ i, t \qquad (2\text{-}8)$$

$$0 < \alpha_{iat} < 1 \qquad 0 \leq \beta_{it} < 1 \qquad \forall\ i,a,t \qquad (2\text{-}9)$$

In this system, parameters are denoted as greek letters and variables as roman letters.[2] Variables and parameters are defined as:

x_{iat} = number of salmon of type i and age a caught in period t,

y_{iat} = number of salmon of type i and age a that are not caught in period t,

α_{iat} = the natural survival rate of uncaught salmon of type i and age a from period to to period t+1,

β_{it} = the survival rate to period t+l of salmon of type i and 1 caught and returned to the fishery in period t,

τ_{iat} = the proportion of salmon of type i and age a that return to the river in period t,

μ_{it} = the number of salmon of type i recruited into the fishery on date t.

Note that one of these systems would exist for each i is contained in I and for each t included in the planning horizon that represents a date when salmon are recruited into the fishery.

System (2-2)-(2-9) is called an accounting system because it does not contain growth or mortality equations. Instead, it simply accounts for the number of salmon of type in the fishery at each of the dates in the interval. Equation (2-2), for example, states that the number of salmon caught in period t plus the number not caught equals the number recruited at the beginning of period t. The adjustment for natural and shaker mortality occurring in period t is accomplished in equation (2-3), which relates the number of salmon surviving at the end of the period to the number caught and not caught in period t+l. In equation (2-3), and in every succeeding equation, the natural and shaker mortality taking place during the time period is accumulated and charged to the fishery's population at the end of the period. As equations (2-5) and (2-6) indicate, the same is true for the number of two year old and older salmon returning to the river in period L. Natural survival rates (α_{iat}), shaker survival rates (β_{it}), and escapement rates (τ_{iat}) are therefore expected values for the period. This type of approximation to the continuous biological processes of the fishery is a result of the treatment of time as a discrete variable.

If the a, β, and τ components are allowed to be functions of the catch and population levels, equations (2-2)-(2-9) would be a relatively general population dynamics system.[3] But this system would also be nonlinear in variables and would rule out linear and quadratic programming formulations. the linear system is obtained by assuming that a, β, and t are predetermined parameters. Thus, this system cannot allow for population density dependent survival and escapement rates.

The particular restrictions included in system (2-2)-(2-9) maintain that all seven year old salmon return to the river, that some "jacks" may return at ages three, four, or five, and that no one year old salmon return. Salmon of age three to five are not required to return: note, for example, that setting $\tau_{iat}=0$ would imply that no two year old salmon of type i return to the river in period t. It should also be noted that setting $\tau_{iat}=1$ will imply that all surviving salmon have returned to the river by age a. The variables x and y in the system for all higher ages will be forced to zero by this parameter choice.

The population dynamics system also incorporates the policy that salmon caught during their first year in the ocean are returned to the fishery. This management policy may be removed by setting $\beta_{it}=0$ for all i and t. It should be noted that, as defined, β_{it} measures both the immediate mortality from the trauma of being hooked and the natural mortality that would occur during period t.

Although simple in concept, system (2-2)-(2-9) is complicated by the need to keep track of independently-varying and time period subscripts. A special case in which all time periods are set equal to one year and only one cohort is assumed to enter the fishery in each year greatly simplifies the algebra. Yet this special case exhibits all of the essential aspects of the more general system. We shall introduce this special system here and use it in the ensuing development. The procedure necessary to develop the more general system should be obvious from the special case.

The single-cohort annual period system is

$$x_{i1t} + y_{i1t} = \mu_{it} \tag{2-10}$$

$$\alpha_{i1} y_{i1t} + \beta_i x_{i1t} = x_{i2,t+1} + y_{i2,t+1} \tag{2-11}$$

$$\alpha_{ia}(1 - \tau_{ia}) y_{ia,t+a-1} = x_{i,a+1,t+a} + y_{i,a+1,t+a} \qquad a = 2,\ldots,5 \tag{2-12}$$

where all variables and parameters are as earlier defined. Note that this system also incorporates the assumption that the mortality and escapement rates are time invariant. Thus, the time subscript is suppressed on the a, β, and τ parameters.

FOCAL INTERVALS AND CONSTRAINT SETS

It would be impossible to fish for a single cohort or age of salmon in a fishery where several cohorts are comingled. Thus, we need to expand the cohort equation system of the last section to account for the catch of all ages of salmon in a particular time period. We also need to add a set of escapement goals to complete the constraint set for a given type of salmon. The result is a population dynamics “module” which can be used as a basic building block in a mathematical programming model. One module for each type of salmon present in the fishery will form a basic constraint set for a model of the fishery. This set can be revised as needed to represent different management controls or to fit different model forms. It is the “heart” of the programming models under consideration.

To model fishing mortality for a given type of salmon in a given time period, it becomes necessary to choose a focal cohort and to consider all other cohorts of this type of fish simultaneously present in the fishery. If recruitment and escapement are to be related through the population dynamics of the focal cohort, enough time periods must be considered to account for the tenure of the cohort in the fishery. Under the assumptions of the special case described by the equation system (2-10)-(2-12), six time periods would be needed. For the general system (2-2) - (9), t_7-t_1 time periods would be required. These periods constitute the focal interval of the constraint set.

By definition, only the tenure of the focal cohort of salmon will fall completely within the focal interval. All other cohorts simultaneously present with this cohort will have one or more periods that fall outside the interval. This is illustrated in figure 1 for the cohort system (2-10)-(2-12). Note that if the model has the focal interval as its planning horizon, initial conditions will have to be established for cohorts 0, -1, .., -4 and terminal conditions for cohorts 2, 3, ..., 6. Mortality occurring prior to the focal interval can be used to develop the initial conditions. Minimum restrictions on the number of salmon remaining in the fishery in the period after the focal interval can be used to terminate the model.

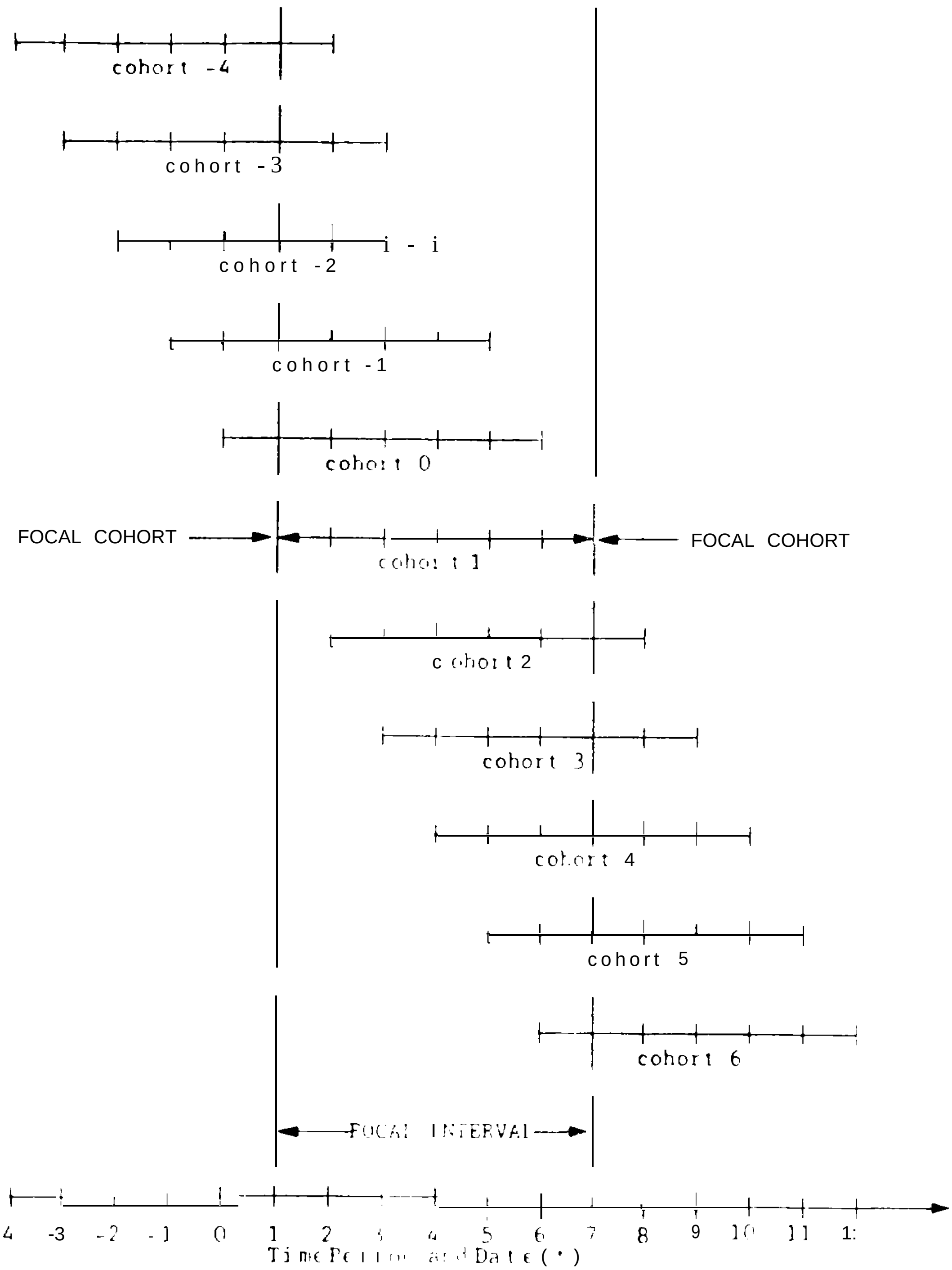
Let the numbers of salmon of type i remaining in the fishery in the period after the focal interval be denoted by $w_{i2}, w_{i3}, \ldots, w_{i6}$, where w_{i2} is the number of two-year old salmon, wi3 is the number of three-year old salmon, and so on. Then if minimum population levels are specified for salmon of ages two to six in the seventh period, the terminal conditions become :

$$W_{ia} \geq \tilde{w}_{ia} \quad a = 2,\ldots,6 \qquad i \in 1 \tag{2-13}$$

where w_{ia} is the predetermined minimum population level. These restrictions may be effectively removed by setting W_{ia} equal to zero. but it is unlikely such a management policy would be endorsed, since it would imply possible extinction for the salmon.

The equation system for the focal interval corresponding to the case represented in figure 2.1 is:

Figure 2.1. Representation of Cohorts of Salmon of Type i Present in Fishery During Tenure of the Focal Cohort. Assumptions Include One Cohort Per Year and Annual Time Periods



$$x_{ia1} + y_{ia1} = b_{i,\,2-a} \qquad a=2,\ldots,6 \qquad (2\text{-}14)$$

$$x_{i1t} + y_{i1t} = \mu_{it} \qquad t=1,\ldots,6 \qquad (2\text{-}15)$$

$$-\alpha_{i1}y_{i1t} - \beta_i x_{i1t} + x_{i2,t+1} + y_{i2,t+1} = 0 \qquad t=1,\ldots,5 \qquad (2\text{-}16)$$

$$-\alpha_{ia}(1-\tau_{iat})y_{iat} + x_{i,a+1\,t+1} + y_{i\,a+1\,t+1} = 0 \quad a=2,\ldots,5,\ t=1,\ldots,5 \qquad (2\text{-}17)$$

$$\alpha_{i1}y_{i16} + \beta_i x_{i16} \geq \tilde{w}_{i2} \qquad (2\text{-}18)$$

$$\alpha_{ia}(1-\tau_{ia})y_{ia6} \geq \tilde{w}_{i,a+1} \qquad a=2,\ldots,5 \qquad (2\text{-}19)$$

$$\sum_{a=2}^{6} \tau_{ia}y_{iat} \geq \delta_{it} \qquad t=1,\ldots,6 \qquad (2\text{-}20)$$

Equations (2-14) – (2-19) are obtained by repeatedly applying the cohort system (2-10) – (2-12) to the eleven cohorts of the focal interval. These equation are rearranged so that the initial conditions for cohorts 0, –1,...,4 are given in equation (2-14) and the terminal conditions for cohorts 2,...,6 are given in (2-18) and (2-19). Equation (2-20) introduces escapement goal constraints which state that the number of salmon of type i returning to the river during period t must equal or exceed the goal δ_{it}. All variables and parameters except b_{it} are as defined previously; b_{it} is defined below.

The terminal equations (2-18) and (2-19) are obtained by first applying the definition of wit to the cohort system (2-10) – (2-12) for the cohorts 2, ..., 6 and then substituting into (2-13). The variables $b_{i,2-a}$ represent the number of salmon from cohorts 0, –1,...,-4 present in the fishery on the beginning date of the focal interval. These coefficients may be computed from historical catch statistics by applying the following formulas:

$$b_{i,-4} = \prod_{a=2}^{5} \alpha_{ia}(1-\tau_{ia})[\alpha_{i1}\mu_{i,-4} - (\alpha_{i1}-\beta_i)\tilde{x}_{i1,-4} - \tilde{x}_{i2,-3}] \qquad (2\text{-}21)$$

$$- \prod_{a=3}^{5} \alpha_{ia}(1-\tau_{ia})\tilde{x}_{i3,-2} - \prod_{a=4}^{5}\alpha_{ia}(1-\tau_{ia})\tilde{x}_{i4,-1} - \alpha_3(1-\tau_3)\tilde{x}_{i50}$$

$$b_{i,-3} = \prod_{a=2}^{4} \alpha_{ia}(1-\tau_{ia})[\alpha_{i1}\mu_{i,-3} - (\alpha_{i1}-\beta_i)\tilde{x}_{i1,-3} - \tilde{x}_{i2,-2}] \qquad (2\text{-}22)$$

$$-\prod_{a=3}^{4}\alpha_{ia}(1-\tau_{ia})\tilde{x}_{i3,-1} - \alpha_{i4}(1-\tau_{i4})\tilde{x}_{i40}$$

$$b_{i,-2} = \prod_{a=2}^{3}\alpha_{ia}(1-\tau_{ia})[a_{i1}\mu_{i,-2} - (\alpha_{i1}-\beta_{i})\tilde{x}_{i1,-2} - x_{i2,-1}] \tag{2-23}$$

$$- \alpha_{i3}(1-\tau_{i3})\tilde{x}_{i30}$$

$$b_{i,-1} = \alpha_{i2}(1-\tau_{i2})[\alpha_{i1}\mu_{i1,-1} - (\alpha_{i1}-\beta_{i})\tilde{x}_{i1,-1}] \quad \alpha_{i2}(1-\tau_{i2})\tilde{x}_{i20} \tag{2-24}$$

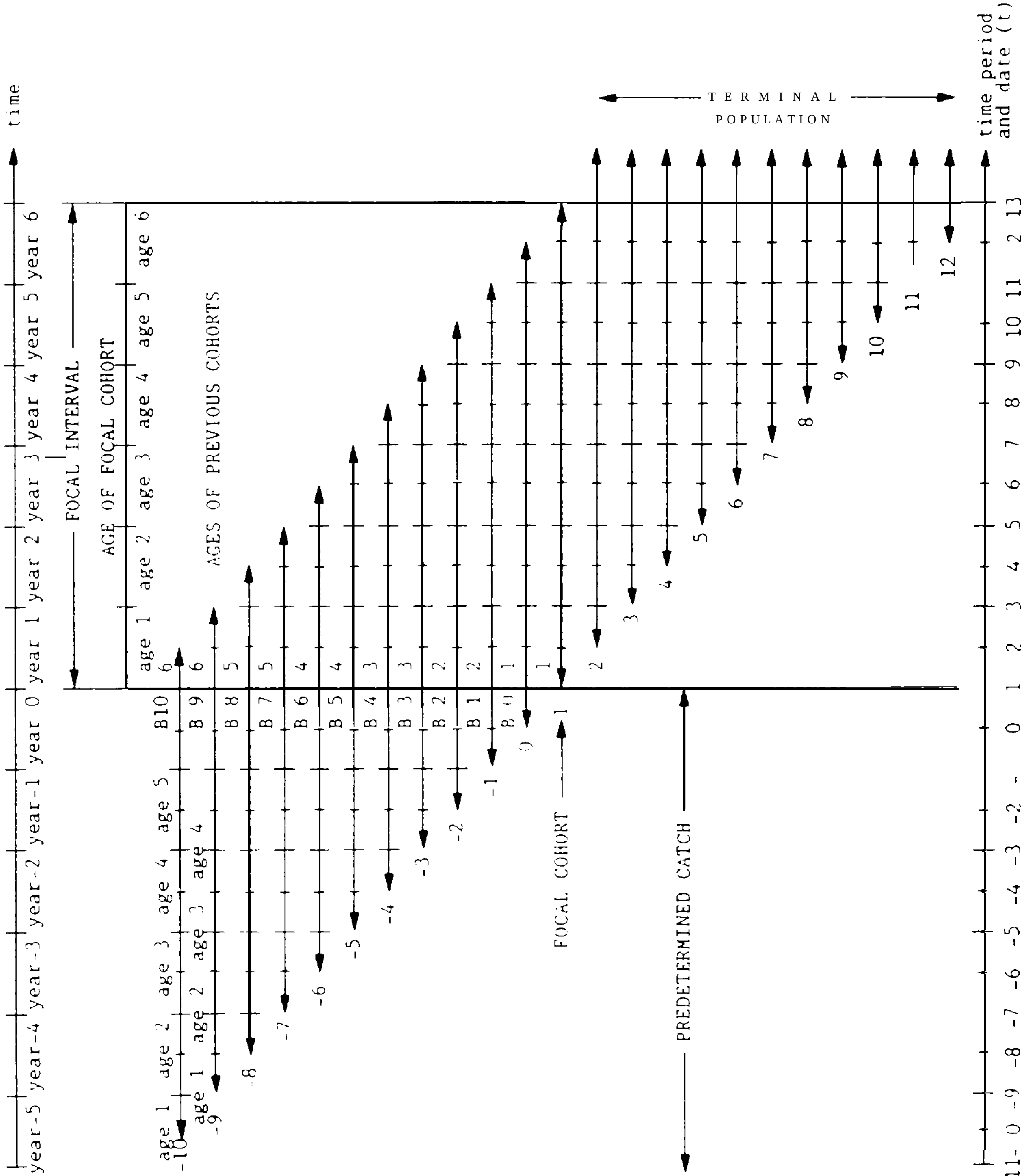
$$b_{i0} = \alpha_{i1}\mu_{i10} - (\alpha_{i1}-\beta_{i})\tilde{x}_{i10} \, . \tag{2-25}$$

Formulas (2-21) – (2-25) are derived from the cohort equation systems by substituting out the y_{iat} population variables for the time periods 0, -1,...,-4. The bar over the catch variables xiat indicate that these variables should be assigned their historical values for the time periods preceeding t=1. Since these values would be known, the computed coefficients for the b_{it} would provide the initial conditions needed to model the population dynamics within the focal interval.[4]

The population dynamics module illustrated by (2-14) – (2-20) may be stated for a more general situation by performing similar algebraic manipulations on the cohort system (2-2) – (2-9). Although the procedure is the same, the algebra for this general case is more tedious, and the number of equations is significantly larger. Steps for constructing such a population dynamics module are given in Appendix 2A. It should be noted, however, that an explicit set of equations and formulas for computing the initial conditions cannot be written until a particular set of dates and periods is established.

Figure 2.2 illustrates how both the number of equations for- a cohort and the number of cohorts in the focal inter-val increase as the number of defined time periods is increased. This figure is based on the assumption that two cohorts of salmon enter in each year and that all time periods are six months in length. The figure shows that the number of cohorts appearing in the

Figure 2.2. Representation of Cohorts of Salmon: Two Cohorts Per Year, Semi-Annual Time Periods



focal interval increases from 11 to 23 and that the number of equations required to describe the population dynamics increases from 47 to 178 (one equation for each point in the focal interval plus one escapement goal for each cohort ending in the interval).[5] This number would decrease to 172 if an escapement goal is established for each year instead of each cohort. If, say, 15 types of salmon are identified in the fishery, the linear constraint set of this mathematical programming model would consist of 2,670 equations. The tendency for the constraint set to grow big as more realistic models are formulated justifies our earlier concern about solution method and model form. It also explains why linear programming might be the most feasible of the various mathematical programs that can be constructed around the linear constraint set for the ocean salmon fishery.

MODELS WITH LINEAR CONSTRAINT SETS

The population dynamics module formulated in the previous section offers no way to choose particular values of the x_{iat} as either more desirable or more feasible. To model these aspects of fishing mortality, we need to represent catch in a way that makes it responsive to the effects of management policies, technical realities and fishing practices. The definition of desirability and the determination of desirable catch levels is the role of the objective function in mathematical programming. Establishing what can be feasibly caught from the subpopulations requires further development of the constraint set.

The missing ingredients would be supplied in a linear programming model through the introduction of a linear objective function and a set of linear fishing activities. Since linear programming is one of the least restrictive mathematical programming formulations in terms of the number of feasible variables and equations, we shall begin with this modelling alternative. The linear objective function and fishing activity will be introduced first. These components then will be combined with the example formulation of the population dynamics module presented in equations (2-14) - (2-20) to produce an illustrative lineal- programming model of the ocean salmon fishery. TO simplify exposition, the more general model based on the cohort system (2-2)

– (Z-9) will not presented. It can be constructed, however, in the same way that the illustrative model is developed.

Linear Objective Functions

One simple linear objective function would be to maximize the total catch from the salmon fishery:

$$\text{Maximize} \quad x = \sum_t \sum_i \sum_a x_{iat} \tag{2-26}$$

Under this criterion, escapement goals would be exceeded only because fishing cannot exactly target particular ages and types of salmon. This criterion takes no explicit cognizance of either the demand for salmon or the costs of harvest. It would be economically efficient only if demand is essentially perfectly elastic at prevailing prices, and the fishing industry is over-capitalized to the extent that virtually any amount of salmon can be caught at a constant marginal cost. Then, if marginal cost is less than price for all types and ages of salmon, the maximum catch criterion would correspond to the maximum profit criterion commonly used in economic analysis.

Since maximum profit typically refers to an individual producer and the models being considered are for the whole fishery, the appropriate analogue to this criterion is to maximize the total net value of the catch:

$$\text{Maximize} \quad p = \sum_t \sum_i \sum_a (p_{iat} - c_{iat})x_{iat} \tag{2-27}$$

The variable p_{iat} represents the price received per fish, and the variable c_{iat} represents the per-fish cost of harvest for a salmon of type i and age a in period t. If this function is to be linear in variables and parameters, prices and average costs must be fixed and predetermined for a given type and age of salmon. These parameterized variables may also be expressed in per-pound units if the x_{iat} are converted from numbers of fish to weight. This option will not be developed here.

Comparison of (2-26) and (2-27) shows the maximum catch criterion to be a special case of the maximum net value of harvest criterion. When (2-27) is used, priority will go to catching the most profitable salmon. When (Z-26) is used, all salmon receive the same priority. In neither case, however, can the quantities caught affect the prices received or average costs expended. Thus these objective functions are quite restrictive with respect to the assumptions made about demand and supply. Quadratic and separable programming offer two ways to generalize the form of the objective function so that price and average harvest costs can vary with quantity caught. These alternatives will be considered after the linear programming model is completed.

Linear Fishing Activities

Let us now consider the missing component in the constraint set. Evidence that something is omitted from this set can be easily found by noting that the population dynamics modules are independent over the types of salmon. Harvest obviously is not, since a decision to fish for one type of salmon will lead to an increased catch of the other commingled types. The relationship between types of salmon needs to be specified before the constraint set will embody all of the population dynamics of the fishery.

One way to do this is through a linear programming activity. Define, for each time period, a vector of catch proportions:

$$\pi = [\pi_{11t}, \ldots, \pi_{16t}, \pi_{21t}, \ldots, \pi_{26t}, \ldots, \pi_{n1t}, \ldots, \pi_{n6t}]$$

where the components of this vector extend over $i = 1, \ldots, n$, and n is the number of types of salmon from the set I. The components of π are defined by:

$$\pi_{iat} = x_{iat}/x_t \qquad \text{where } x_t = \sum_i \sum_a x_{iat}$$

Thus, x_t is the total catch in period t. Now assume the composition of the catch can be pre-specified so that the π_{iat} are predetermined parameters and substitute $\pi_{iat}x_t$ for the catch variables x_{iat} in the population dynamics modules. Then as total catch (x_t) is increased or decreased, the numbers of salmon remaining in the subpopulations (y_{iat}) simultaneously increase or decrease for all i. This simultaneity supplies the missing ingredient in the population dynamics constraint set, for it introduces the effect of fishing mortality on the total population of the fishery.

The substitution of $\pi_{iat}x_t$ for x_{iat} introduces one linear programming activity for each time period into the linear constraint set. (Note that the predetermined π_{iat} become the input-output coefficients, and x_t becomes the level of this activity.) Use of a single activity is highly restrictive for it implies that only one fishing method is used in the fishery in each period. Thus one would want to define a set of fishing activities that are the same for all periods but that allow variation in fishing method within each period. This would imply that fishing methods do not evolve over time (i.e., constant technology) but that fishermen are responsive to changes in the size of the salmon population and to the distribution of salmon throughout the geographic area of the fishery. It would also allow controls on fishing mortality that operate through controls on effort or intensity to be included in the salmon fishery model.

Use of the same set of fishing activities over time is proposed because it provides a basis for estimating values for the π_{iat} parameters.[6] To see this, let us temporarily return to the assumption of a single fishing activity and add the condition that this activity be the same over all time periods. Then catch proportions that are observed over time may be interpreted as random drawings from an underlying multivariate distribution of proportions that remains invariant over time and over age and type of salmon.

Expected proportions could then be estimated from the historical catch data and used in the linear programming model of the ocean fishery. By definition,

$$\begin{bmatrix} x_{i11} & \cdots & x_{i1t} \\ \vdots & & \vdots \\ x_{i61} & \cdots & x_{i6t} \end{bmatrix} = \begin{bmatrix} \pi_{i1t} \\ \vdots \\ \pi_{i6t} \end{bmatrix} [x_1 \cdots x_t] \qquad i = 1, \ldots, n \tag{2-28}$$

where the x_{iat}, x_t, and π_{iat} are historical observations. By the above assumptions, the expected proportions are constant over time:

$$\hat{\pi}_{ia} = \hat{\pi}_{ia1} = \hat{\pi}_{ia2} = \cdots = \hat{\pi}_{iat}$$

where "hat" denotes expected value. Upon substitution of expected for observed proportions (2-28) becomes:

$$\begin{bmatrix} x_{i11} & \cdots & x_{i1t} \\ \vdots & & \vdots \\ x_{i61} & \cdots & x_{i6t} \end{bmatrix} \begin{bmatrix} 1/x_1 \\ \vdots \\ 1/x_t \end{bmatrix} = t \begin{bmatrix} \pi_{i1} \\ \vdots \\ \pi_{i6} \end{bmatrix} \tag{2-29}$$

and

$$\pi_{ia} = \frac{1}{t} \sum_t (x_{iat}/x_t). \tag{2-30}$$

Thus, the expected proportions would be simple averages of the historical data.

The introduction of a set of time invariant fishing activities is consistent with replacement of the multivariate distribution by a set of conditional distributions whose parameters depend on the given fishing methods. Expected catch proportions would be estimated in this case either by subdividing the historical catch data according to fishing method and using simple averages for each subgroup, or by regressing catch proportions against total catch and measures of effort and intensity. Either procedure would be feasible provided the constant technology and time invariant distribution assumptions are acceptable.

A Linear Programming Model

The necessary components are now available for a linear programming model of the ocean salmon fishery. All that remains is to assemble them. To obtain the requisite set of linear fishing activities, we replace the single activity π by:

$$\pi_k = [\pi_{i1k},\ldots,\pi_{i6k},\pi_{21k},\ldots,\pi_{26k},\ldots,\pi_{I1k},\ldots,\pi_{I6k}]$$

where k is an element of the set K of all fishing activities. Total catch becomes :

$$x_t = \sum_k \pi_k x_{kt} \qquad k = 1,\ldots,m \tag{2-31}$$

where x_{kt} is the total catch of salmon in period t using fishing method k, and m is the number of methods included in the model. Quantities of salmon caught of type i and age a in period t become:

$$X_{iat} = \sum_k \pi_{iak} x_{kt} \tag{2-32}$$

Thus, the linear fishing activities may be introduced into the population dynamics modules by substituting $\sum_k \pi_{iak} x_{kt}$ for the x_{iat}.

Substitution of the fishing activities for the catch variables in the population dynamics modules yields the basic constraint set for the linear programming model. To this constraint set is added the objective function for maximizing the net value of the catch and the non-negativity conditions for the variables. After these substitutions, the illustrative linear programming model for the single-cohort annual-period example may be stated as:

Maximize:

$$\sum_t \sum_i \sum_a \sum_k (\tilde{p}_{iat} - \tilde{c}_{iat}) \pi_{iak} x_{kt} \tag{2-33}$$

subject to

$$y_{ia1} + \sum_k \pi_{iak} x_{k1} = \tilde{b}_{i,2-a} \qquad a = 2,\ldots,6 \quad i = 1,\ldots,n \quad (2\text{-}34)$$

$$y_{i1t} + \sum_k \pi_{i1k} x_{kt} = \mu_{it} \qquad t = 1,\ldots,6 \quad i = 1,\ldots,n \quad (2\text{-}35)$$

$$-\alpha_{i1} y_{i1t} - \beta_i \sum_k \hat{\pi}_{i1k} x_{kt} + \sum_k \hat{\pi}_{i2k} x_{k,t+1} + y_{i2,t+1} = 0 \qquad (2\text{-}36)$$
$$t = 1,\ldots,5 \quad i = 1,\ldots,n$$

$$-\alpha_{ia}(1 - \tau_{ia}) y_{iat} + \sum_k \hat{\pi}_{i,a+1,k} x_{k,t+1} + y_{i,a+1,t+1} = 0 \qquad (2\text{-}37)$$
$$a = 2,\ldots,5 \quad t = 1,\ldots,5 \quad i = 1,\ldots,n$$

$$\alpha_{i1} y_{i16} + \beta_i \sum_k \pi_{i1k} x_{k6} \geq \tilde{w}_{i2} \qquad i = 1,\ldots,n \quad (2\text{-}38)$$

$$\alpha_{ia}(1 - \tau_{ia}) y_{ia6} \geq \tilde{w}_{i+g',a+1} \qquad a = 2,\ldots,5 \quad i = 1,\ldots,n \quad (2\text{-}39)$$

$$\sum_{a=2}^{6} \tau_{ia} y_{iat} \geq \delta_{it} \qquad t = 1,\ldots,6 \quad i = 1,\ldots,n \quad (2\text{-}40)$$

and to:

$$Y_{iat} \geq 0 \ \forall \ i,a,t \qquad x_{kt} \geq 0 \ \forall \ k,t \qquad \tilde{w}_{ia} \geq 0 \ \forall \ i,a \qquad (2\text{-}41)$$

That the linear programming model is simply an assembly of earlier discussed components can be seen by comparing equations (2-33) with (2-27) and equations (2-34) – (2-40) with (2-14) – (2-20). The objective function comparison shows that net value is computed using fixed prices and costs for all of the types of salmon present in the fishery and that catch of a particular type and age of salmon in a given time period is computed as a weighted average of the salmon caught by different fishing methods in that period. The constraint comparison shows that the full constraint set is composed of n population dynamics modules which are linked together by the fishing activities. Since $\Sigma\pi_{iak}x_{kt} = x_{iat}$, the interpretation of the individual constraints remains exactly as in the cohort system from which they are derived. Recruitment for each cohort is still divided into salmon that are caught and not caught. Predetermined natural survival, shaker survival and escapement rates are still applied to l-educe salmon populations when the fish are transferred from period t to period t+1.

Initial conditions for the focal interval are also still calculated from historical catch data and applied to cohorts entering the fishery before the beginning of the focal interval. Escapement goals are also applied as before. Because of the variable substitution, however, the linear programming model is expressed in terms of x_{kt} instead of x_{iat}. Thus the non-negativity conditions originally stated in equation (Z-7) are revised in equation (2-41).

Two modifications may be easily incorporated into this model. Setting all of the $\tilde{p}iat$ – Eiat coefficients equal to one will convert the net value objective function (2-33) to a function that maximizes total catch. Controls on fishing mortality resulting from controls on fishing method can be modeled by adding constraints of the form:

$$\sum_{k} \Gamma_k x_{kt} \leq \tilde{x}_{kt} \qquad \Gamma k \ \varepsilon(0,1) \qquad (2\text{-}42)$$

where $\tilde{x}_{kt}$ is a predetermined maximum catch level for period t.

Nonlinear Extensions and Conclusions

It was noted earlier that the assumption of fixed prices and costs is a stringent assumption of the linear programming model. This is especially so since there appears to be substantial agreement that the appropriate criterion is to manage the fishery so as to maximize its economic surplus (U.S. Department of Commerce, 1984). A detailed discussion of possible procedures for valuing changes in salmon stocks is presented in the next chapter. Discussion here is limited to how the surplus criterion might be incorporated into a mathematical programming model based on the defined linear set of population dynamics constraints.

Samuelson long ago showed that a quadratic objective function formed by substituting price-dependent demand and supply curves for the fixed prices and costs of equation (2-27) could be interpreted as an economic surplus criterion in which the sum of producers and consumers surplus is maximized.

Thus, one can introduce the preferred criterion by substituting the objective function

$$S = \sum_t \sum_i \sum_a [(\phi_{oiat} - \phi_{1iat}x_{iat}) - .5(\Omega_{0iat} + \Omega_{1iat}x_{iat})]x_{iat} \qquad (2\text{-}43)$$

for the linear objective function (Z-27). This function contains the linear demand and supply curves:

$$p_{iat} = \phi_{0iat} - \phi_{1iat}x_{iat} \quad i = 1,...,n \quad a = 1,...,6 \quad t = 1,...,6 \qquad (2\text{-}44)$$

$$c_{iat} = \Omega_{0iat} + \Omega_{1iat}x_{iat} \quad i = 1,...,n \quad a = 1,...,6 \quad t = 1,...,6 \qquad (Z\text{-}45)$$

where the ϕ and Ω_1 are non-negative parameters, and Ω_0 is a positive or negative parameter. Simple linear cross-price demand and supply functions may also be incorporated into the quadratic objective function, provided certain symmetry conditions are met (McCarl and Spreen, Takayama and Judge).

Formulation of a quadratic function could also offer an opportunity to decompose catch into commercial and sport segments and to formulate separate demand functions for these separate salmon markets. Each x_{iat} would be defined as the sum of $x^c_{iat} + x^s_{iat}$ in the objective function, where x^c represents the number of salmon caught commercially, and x^s represents the number of salmon caught by sports fishermen. The set of linear constraints defined for the linear programming model could be used as given in this case if the following constraints are added to the set:

$$\sum_{k=1}^{m} \pi_{iak}x_{kt} - x^c_{iat} - x^s_{iat} = 0 \quad i = 1,...,n \quad a,t = 1,...,6 \qquad (2\text{-}46)$$

Alternatively, separate sets of fishing activities could be defined for each group of fishermen and the x_{kt} decomposed into x^c_{kt} and x^s_{kt}. Additional constraints would not be required in this case, and it would allow different costs of harvest to be specified for the commercial and sports fishing activities.

Separable programming models also offer the opportunity to include the economic surplus criterion in a mathematical programming model of the ocean salmon fishery . Such models would not be restricted to linear demand and supply functions. The linear objective function would instead be replaced with any separable objective function of the form:

$$s = \sum_t \sum_i \sum_a f_{iat}(x_{iat}) \cdot x_{iat} \qquad i = 1, \ldots, n \quad a, t, = 1, \ldots, 6 \qquad (2\text{-}47)$$

where the f are single-valued continuous functions (Hadley). Against this advantage, however, is the disadvantage that separable programming models require a variable transformation that causes them to grow very large when the number of separable variables is large. This type of model might be successfully formulated and solved for a realistic number of types of salmon and a useful set of fishing activities if it is convex. But we suspect the fishery model would be non-convex, since there is no reason to believe that all of the own-price demand functions will be concave to the origin. Given the size of the linear constraint set, we do not believe a non-convex separable programming model will be feasible for the ocean salmon fishery.

A quadratic programming model may also suffer from a size problem, since the dimensions of the linear problem that must be solved roughly doubles when a quadratic objective function is specified (Takayama and Judge). Thus the nonlinear extensions of the linear programming fishery model will limit the number of time periods and types of salmon included in the analysis. There may also be a more fundamental problem with models based on the linear constraint set. This potential problem arises from the use of the linear fishing activity to tie together the population dynamics modules.

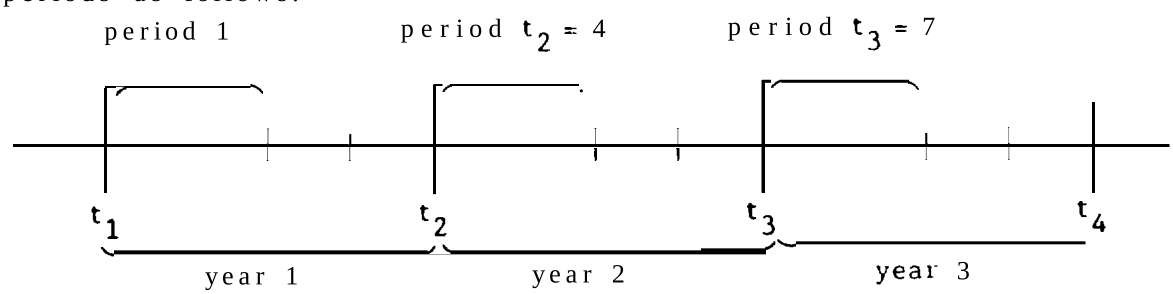
The latter aspect is explored in some detail in the appendix to this chapter and the possibilities for development of a nonlinear model are examined. Whether one can stick with the relative simplicity of a linear model or whether it will be necessary to resort to nonlinear formulations turns largely on empirical questions. For example, whether the Columbia based commercial fishery is a sufficiently large share of the mar-ket so that variations in catch will significantly affect price, and whether- natural mortality differs substantially among species and stocks and, if so, whether data can be collected to reflect these differences. The appendix demonstrates, we believe, that a nonlinear model is feasible, should it be required.

In any case we believe, based on the Phase II work, that mathematical programming offers an attractive framework for organizing and conducting economic analysis in the ocean fisheries area. We believe this line of work should be pursued in Phase III and an early effort be made to resolve the empirical questions that bear on the specific form of model to be used.

We turn in the next chapter to the problem of establishing economic values for the various activities included in a programming model.

ENDNOTES

1. This scheme may be illustrated for one six-month and two quarterly periods as follows:



Note that period t is the period of time immediately following the date t in this scheme, and that the cohort is recruited at date $t=t_1=1$.

2. This convention will be maintained throughout the chapter. If a variable is assigned a predetermined value, it will be marked by a bar over the variable.

3. It would not be completely general because time is treated as discrete when it is, in fact, continuous.

4. The formulas obtained by substituting the y_{iat} variable out of the population dynamics equations may also provide a means to estimate the predetermined survival and escapement rates from historical catch data. This might be accomplished by applying the ARIMA time series technique to the relationship:

$$x_{i6t} = \prod_{a=1}^{5} \alpha_{ia}(t - \tau_{ia})\mu_{i,\,t-5} - y_{i6t}$$

$$- \prod_{a=2}^{5} \alpha_{ia}(1 - \tau_{ia})(\alpha_i - \beta)x_{i1,\,t-5} \prod_{a-2}^{5} \alpha_{ia}(1 - \tau_{ia})x_{i2,t-4}$$

$$- \prod_{a=3}^{5} \alpha_{ia}(1 - \tau_{ia})x_{i3,\,t-3} - \prod_{a=4}^{5} \alpha_{ia}(1 - \tau_{ia})x_{i4,\,t-2}$$

$$- \alpha_{i5}(1 - \tau_{i5})x_{i5,t-1}.$$

This relationship is a six-period autoregressive process that meets the fundamental criterion for stationarity (mccleary and hay). thus it could form the foundation for an econometric estimation of the $\alpha_{ia}(1 - \tau_{ia})$,

a_{11} and β_i parameters. It would not be able to separate estimates for çia from those for α_{ia}, however, and the estimation would require a long series of detailed catch data.

5. This number would decrease to 172 if an escapement goal is established for each year instead of each cohort.
6. This assumption can be relaxed if some other way of estimating the π_{iat} can be found.

Appendix to Chapter 2

A Problem in Fish Population Dynamics and a Possible Nonlinear Model

INTRODUCTION

Although the linear constraint set is attractive because of its simplicity, the combination of linear population dynamics system and linear fishing activities may bias the catch toward the salmon with higher natural mortality rates. This possible bias can be shown by an illustrative example. Let us suppose that there are 110 fish of each of two types of salmon in the fishery at the start of period t. During this period fishing mortality claims ten salmon of each type, while natural mortality claims ten salmon of type 1 and 20 salmon of type 2. (In the model's notation $x_t=20$, $\pi_{1t}=\pi_{2t}=.5$, $\alpha_{1t}=.9$, $\alpha_{2t}=.8$, $\tau_{1t}=\tau_{2t}=0$. Note that a is suppressed.) Now suppose that total catch from this cohort is 16 salmon in period t+l. According to the formulation of the linear fishing activity, this catch would decompose into 8 salmon of each type. But because 90 salmon of the first type and 80 of the second type remain in the fishery, the decomposition implies that 8 out of 90 or 8.8 percent of the type 1 salmon succumb to fishing mortality while 8 out of 80 or 10 percent of the type 2 salmon are caught. Thus the use of the linear fishing activity biases the proportion of the catch in successive periods towards the salmon with the higher natural mortality.

What prompts concern in this example is the inherent belief that if the fishing is done in the same way and at the same effort and intensity over time, then the probability that a particular salmon will be caught should remain the same. Such a constant probability is incorporated into the linear fishing activity. But it is not respected by the linear constraint set containing the activity. The result is an inherent conflict in the embodied assumptions about fishing mortality.

In the following we shall first introduce an equal or predetermined probability-of-catch assumption which can be used in place of the linear fishing activities. Then the linear constraint set will be reformulated, a nonlinear model stated, and a solution procedure discussed. The nonlinear model removes the potential bias in the catch proportions. [Since much of the notation used in this appendix is introduced in the chapter proper, equations are numbered continuously with those in the text.]

A Probability-of-Catch Assumption

To state this assumption, we first need to define some aggregate catch and population variables. Let:

$x_{it} = \sum_a x_{iat}$ = catch of salmon of type i in period t

$y_{it} = \sum_a y_{iat}$ = subpopulation of salmon of type i that remains uncaught in period t.

$x_t = \sum_i x_{it}$ = total catch in period t

$y_t = \sum_i y_{it}$ = total subpopulation of salmon not subjected to fishing or shaker mortality in period t.

Also, let

$$z_{iat} = x_{iat} + y_{iat}$$

$$z_{it} = x_{it} + y_{it}$$

$$z_t = x_t + y_t$$

be the corresponding measures of total population in the fishery at the start of period t. This notation allows us to define some probabilities that salmon will be caught under the simple assumption that every salmon within a given fishing area has a equal chance of being hooked. Given this equal probability assumption, the probability that a particular salmon will be caught in period t is $1/z_t$. The probability a salmon of type i will be caught is x_{it}/z_t and the probability a salmon of type i and age a will be caught is x_{iat}/z_t. If enough fishing takes place and this assumption is correct, actual catch proportions should approximate these probabilities.

An equal probability of being caught for all salmon in the fishery would imply that:

$$\frac{X_{iat}}{X_t} = \frac{Z_{iat}}{z_t} \tag{2-48}$$

In words, this equation states that the frequency with which a salmon of type i and age a appears in the catch would be equal to the frequency with which

it appears in the population. However, the nature of some fishing practices and fishing management controls is such that the catch proportions of particular types and ages of salmon are greater or smaller than the corresponding population proportions. Fishing may take place near the mouth or in the lower river, for example. This would increase the size (as well as the number) of salmon caught. Fishing location may also be varied to obtain more salmon of a particular species, or larger hooks and different bait employed to discourage smaller salmon from biting. It may also be true that particular types or ages of salmon are more aggressive feeders and thus more liable to fishing mortality.

These cases may be described by extending the equal probability-of-catch assumption into a predetermined probability-of-catch assumption. Define:

$$\frac{X_{iat}}{X_t} = \sigma_{iat} \frac{Z_{iat}}{z_t} \tag{2-49}$$

where $\sigma_{iat} \leq 0$ is the ratio of the catch proportion x_{iat}/x_t to the population proportion z_{iat}/z_t. From a descriptive viewpoint, (Z-49) is a tautology. But it provides a way to define predetermined parameters that establish expected probabilities of catch for the fishery model. The predetermined $\hat{\sigma}_{ia}$ may reflect historical averages (as argued for the π_{ia} earlier) or they may reflect proposed policies aimed at affecting the distribution of the catch with respect to age and type. Some algebra will show that

$$\sum_i \sum_a \sigma_{iat} \frac{z_{iat}}{z_t} = 1 \qquad i = 1, \ldots, n \quad a = 1, \ldots, 6 \tag{2-50}$$

for any time period t. Hence the necessary condition is established for the proportions in (2-49) to be interpreted as probabilities. Equation (49) and its special case (2-48) embody the probability-of-catch assumption of the nonlinear model.

A Nonlinear Constraint Set

Equation (2-48) may be used to demonstrate that the equal probability-of-catch assumption implies:

$$\frac{x_{iat}}{z_{iat}} = \frac{x_{it}}{z_{it}} = \frac{x_t}{z_t} = v_t \tag{2-51}$$

where v_t is the proportion of the total salmon population that is caught in period t. Thus

$$x_{iat} = v_t z_{iat} \tag{2-52}$$

$$y_{iat} = (1 - v_t) z_{iat} \tag{2-53}$$

Similarly, equation (2-49) implies that:

$$x_{iat} = \hat{\sigma}_{ia} v_t x_{at} \tag{2-54}$$

$$y_{iat} = (1 - \hat{\sigma}_{ia} v_t) z_{iat} \tag{2-55}$$

where, for convenience, the predetermined non-negative parameters $\hat{\sigma}_{ia}$ are assumed to be invariant over time.

Relationships (2-54) and (2-55) allow the variables v_t and z_{iat} to be substituted in the linear population dynamics module to create a nonlinear constraint set. To illustrate, the single-cohort annual-period system (2-14) – (2-19) would become:

$$z_{ia1} = b_{i,2-a} \qquad a = 2,\ldots,6 \qquad (2\text{-}56)$$

$$z_{i1t} = \mu_{i,\,t-1} \qquad t = 1,\ldots,6 \qquad (2\text{-}57)$$

$$z_{i2,t+1} = [\alpha_{i1}-(\alpha_{i1} - \beta_i)\hat{\sigma}_{i1}v_t]z_{i1t} \qquad t = 1,\ldots,5 \qquad (2\text{-}58)$$

$$z_{i,a+1,\,t+1} = \alpha_{ia}(1 - \tau_{ia})(1 - \hat{\sigma}_{ia}v_t)z_{iat} \qquad (2\text{-}59)$$

$$a = 2,\ldots,5 \quad t = 1,\ldots,5$$

$$w_{i2} = [\alpha_{i1} - (\alpha_{i1} - \beta_i)\hat{\sigma}_{i1}v_6]z_{i16} \qquad (2\text{-}60)$$

$$w_{i,a+1} = \alpha_{ia}(1 - \tau_{ia})(1 - \hat{\sigma}_{ia}v_6)z_{ia6} \qquad a = 2,\ldots,5 \qquad (2\text{-}61)$$

The equal-probability-of-catch assumption creates the special case of this nonlinear system in which $\hat{\sigma}_{ia} = 1$ for all types and ages of salmon.

The nonlinear constraint system illustrated by equations (2-56) – (2-61) has a special nature which can be exploited in solving the fishery model. It can be seen from inspection of this system that if the total catch proportions v_t can be determined for the focal interval $t=1,\ldots,6$, then the population variables z_{iat} and w_{ia} can be determined by substituting the values for v_t' into the constraint system and solving its equations recursively. Once v_t and z_{iat} are evaluated, the corresponding catch variables x_{iat} and subpopulation variables y_{iat} may be obtained from equations (2-54) and (2-55). Hence the need to simultaneously solve for a large set of variables, which was so evident in the models based on the linear system, is reduced to a need to simultaneously determine six variables. Once these v_t are evaluated, a complete description of the catch and of the salmon population remaining in the fishery is readily available.

This illustrates two major benefits of the probability-of-catch assumption. It removes the potential bias of the linear system by allowing the constraints to be expressed in terms of the total population variables z_{iat}. And it substantially reduces the computational burden involved in solving the fishery model problem. The assumption also allows the objective function of the model to be quite generally stated. In essence, all that is required of this function is that it depends on the catch and/or population variables, that it be concave, and that it be consistent with the predetermined fishing practices and management policies embodied in the σ_{ia}. Otherwise, this function may take any form and include any set of price, cost, income or other market variables. Thus, the objective function of the nonlinear model may be as simple as the maximized catch alternative or as complex as any of the economic surplus measures discussed elsewhere in this report.

A Nonlinear Model

Let us represent the concave objective function for the nonlinear model as:

$$s = f(X,Z,P;\hat{\pounds}) \tag{2-62}$$

where X, Z, P, and $\pounds$ are vectors of variables and parameters. The vector X contains the x_{iat} as elements, Z is a vector of z_{iat}, and P is a vector containing prices, harvest costs, or any other exogenous variables that determine the producer or consumer surplus from the fishery. Thus P may contain income, population, number of vessels, effort measures, etc. The vector $\hat{\pounds}$ has the predetermined proportions $\hat{\sigma}_{ia}$ as its elements. This vector of parameters will serve to introduce the commercial and sport fishing practices and fishing mortality management controls on which the constraint set is based into the formulated objective function. Hence the function f is conditional on the specified $\pounds$.

Given the objective function (2-62), a nonlinear mathematical programming model may be specified for the ocean salmon fishery as follows:

Maximize :

(2-62, repeated) $\quad s = f(X,Z,P,f)$

subject to:

(2-56, repeated) $z_{ia1} = \tilde{b}_{i,2-a} \qquad a = 2,\ldots,6 \qquad i = 1,\ldots,n$

(2-57, repeated) $z_{i1t} = \mu_{it} \qquad t = 1,\ldots,6 \qquad i = 1,\ldots,n$

$$z_{i2,t+1} - [\alpha_{i1} - (\alpha_{i1}-\beta_i)\hat{\sigma}_{i1}v_t]z_{i1t} = 0 \qquad t=1,\ldots,n \quad i=1,\ldots,n \tag{2-63}$$

$$z_{i,a+1,t+1} - \alpha_{ia}(1-\tau_{ia})(1-\hat{\sigma}_{ia}v_t)z_{iat} = 0 \qquad \begin{matrix} a=2,\ldots,5 \\ t=1,\ldots,5 \quad i=1,\ldots,n \end{matrix} \tag{2-64}$$

$$[\alpha_{i1} - (\alpha_{i1} - \beta_i)\hat{\sigma}_{i1}v_6]z_{i16} \geq \tilde{w}_{i2} \qquad i=1,\ldots,n \tag{2-65}$$

$$\alpha_{ia}(1-\tau_{ia})(1-\hat{\sigma}_{ia}v_6)z_{ia6} \geq \tilde{w}_{i,a+1} \qquad a=2,\ldots,5 \quad i=1,\ldots,n \tag{2-66}$$

$$X_{iat} - \hat{\sigma}_{ia}v_t z_{iat} = 0 \qquad a=1,\ldots,6 \quad t=1,\ldots,6 \quad i=1,\ldots,n \tag{2-67}$$

$$y_{iat} - (1-\hat{\sigma}_{ia}v_t)z_{iat} = 0 \qquad a=1,\ldots,6 \quad t=1,\ldots,6 \quad i=1,\ldots,n \tag{2-68}$$

$$\sum_{a=2}^{6} \tau_{iat}y_{iat} \geq *_{it} \qquad t=1,\ldots,6 \quad i=1,\ldots,n \tag{2-69}$$

and to

$$Z_{iat} \geq 0 \qquad 0 \leq v_t < 1 \qquad \tilde{w}_{ia} \geq 0 \qquad \forall \; i,a,t \qquad \text{(Z-70)}$$

This model is based on the single-cohort annual-period population dynamics module illustrated in figure 2-l and described by equations (2-14) - (2-20). Since it is derived from this module by variable transformation, the underlying population dynamics is not changed. However, the individual equations are expressed in terms of total subpopulations instead of salmon caught and not caught. Equations (Z-56) and (2-57) simply state that these subpopulations are initially equal to the number of salmon entering at the beginning of the focal interval or recruited during it. Equations (2-63) and (2-64) the number of salmon of type i and age a that remain from period t after the subpopulation has been reduced by natural mortality, shaker mortality, escapement and fishing mortality. Equations (2-65) and (2-66) reduce the size of the subpopulations in the last period of the focal interval and establish the levels of fish remaining in the salmon fishery after the interval is over.

The equations (2-67) and (2-68) are equations (2-54) and (2-55), repeated as constraints to close the mathematical programming model. Inclusion of (2-68) also allows the escapement goals used in the linear model to be appended to the nonlinear model as equation (2-69). Similar equations for more general models representing more cohorts and shorter time periods can be developed by following the same steps used to construct this model. Note also that any of the previously specified objective functions may be substituted for (2-62) if desired.

Since the linearity required for the constraint sets of the linear and quadratic programming models is not germane to the nonlinear model, survival and escapement rates no longer are required to be predetermined constants. Instead, the constant rates in the system (2-63) - (2-69) may be replaced by the variable rates:

$$r_{iat} = \alpha_{iat}(\alpha_{ia,t-j}, z_{ia,t-j}) \qquad t = 1,\ldots,6 \quad j = 1,\ldots,5 \quad j < t \qquad (2\text{-}71)$$

$$q_{it} = \beta_{it}(\beta_{i,t-j}, z_{ia,t-j}) \qquad t = 1,\ldots,6 \quad j = 1,\ldots,5 \quad j < t \qquad (2\text{-}72)$$

$$e_{iat} = \tau_{iat}(\tau_{ia,t-j}, z_{ia,t-j}) \qquad t = 1,\ldots,6 \quad j = 1,\ldots,5 \quad j < t \qquad (2\text{-}73)$$

where r_{iat} is a variable natural survival rate for salmon of type i and age a in period t, q_{it} is a variable shaker survival rate, and e_{iat} is a variable escapement rate that is dependent on the type and age of salmon in period t. The opportunity to include variable rates in a nonlinear salmon fishery model allows population density to play a role in determining survival and escapement. However, the functions a, β, and τ must be chosen so that the nonlinear constraint set is convex and compact. Otherwise, the suggested solution procedure may fail. This was not a concern with the linear model, since all standard linear and quadratic mathematical programming formulations satisfy the conditions for a global optimum.

A Suggested Solution Procedure

The form of the nonlinear salmon fishery model suggests a solution procedure based on a Fibonacci or Golden Section grid search. These search algorithms are univariate procedures which can be successively applied to a small set of variables. Such applications have worked well when there is no strong interaction between decision variables (Phillips, et al.) and should perform successfully if applied to the v_t. If they do not, more sophisticated search procedures such as the Hooke-Jeeves algorithm can be applied.

The Fibonacci procedure requires a-priori upper bounds on the decision variables, and is facilitated if the variable range is from zero to one. This is, by definition, the feasible range for the v_t. The procedure also depends on a unimodal objective function, which will exist for the specified fishery model if the function f is concave, and the constraint set is convex. Given these conditions, the search procedure should converge to a global optimum.

The Fibonacci and Golden Section search procedures are sequential techniques which successively reduce the interval in which the optimum value of the objective function must lie. Compared to other search techniques, the Fibonacci will yield the minimum-maximum interval of uncertainty after any given number of iterations. The Golden Section search is less efficient but computationally easier. Since the solution must respect the constraint set of the nonlinear fishery model, we propose a modified search procedure based on a Golden Section search. In it, the range of the v_t catch proportions are first reduced to an interval where the constraint set is feasible. Then the standard Golden Section search procedure is applied to these feasible variable ranges to find the optimum ratios of total catch to total population. The procedure terminates whenever the problem is found to be infeasible or the value of the objective function stabilizes over iterations.

A feasible solution is defined as one which satisfies the non-negativity conditions for the x, y and z, and meets the escapement goals, the terminal population conditions, and any other appended constraints such as equation (2-42). The search for such a solution should be quite rapid even for large ocean salmon fishery models, since modern computers can rapidly evaluate many simple recursive equations. Furthermore, this search will not be susceptible to the numerical round-off or truncation error that can occur in solving large linear or quadratic programming models. Convergence is guaranteed by generating a series of grid points that either converge to the optimum solution or to the "no catch" solution in which all of the $v_t = 0$. If this latter point is obtained and the solution is still infeasible, either recruitment is too low or the escapement goals and/or the terminal population requirements are too high.

The objective function enters the computation only when feasibility has been established and the standard Golden Section search procedure is initiated. Each iteration of the search would then produce two grid points in the form of menus of catch and salmon population variables. These catch and population values would be inserted into the objective function and the resulting objective values compared. The comparison would allow one grid point to be discarded and the search algorithm would then produce another menu for comparison. This process would continue until the successive objective values converge to the optimum value. During this procedure, predetermined values would be specified for P, and equation (Z-67) would be used to determine X. Thus the entire solution procedure would have to be repeated if elements of P are to be varied.

Output from a solution would include a full set of catch variables (v and x) and population variables (z and y). An objective value representing producer and consumer surplus would also be produced. This surplus value would be conditional on the prices, costs, fishing practices and management policies used to establish the objective function. It would also depend on the probability-of-catch parameters (a). Surpluses accruing to the resource stock would be evaluated by a shadow price computation. The effect of changes in prices, costs, probabilities of catch and other components of P would be determined by obtaining a series of solutions to the nonlinear ocean salmon fishery model.

Shadow Prices for the Salmon

One of the advantages of the linear programming model is the set of shadow prices produced as part of its solution. Shadow values may also be computed from the nonlinear fishing model. But they must be explicitly sought since they will not be automatically produced by the nonlinear solution procedure. The necessary requirement for computing these shadow values is an optimum solution to the fishery model. One would then change one of the μ_{it}, δ_{it} or $\tilde{w}_{ia}$ parameters by a small amount, re-solve the model, and obtain a new value of s. The difference between this value of s and the original optimal value of s would be the shadow price.

This price would be interpretable as the value of an additional recruited salmon of type i in period t if a μ_{it} is changed. It would be the value to the ocean fishery of lowering the escapement goal in period t by one salmon of type i if a δ_{it} us varied. And it would be the cost of adding one salmon of type i and age a to the terminal population if w_{ia} is changed. All of these values would be with respect to the ocean fishery, since they measure the change in the surplus generated by this component of the total salmon fishery.

Since the number- of parameters that can be varied is large, an extensive set of shadow prices can be obtained. The computational procedure for- obtaining these shadow values can be programmed as part of the solution procedure of the nonlinear model. Even with this added capability, we do not expect it to be difficult to obtain a solution to the nonlinear ocean fishery model.

FINAL COMMENT

Solutions to the proposed mathematical programming model for the ocean salmon fishery should be able to provide answers to most questions about the effect of changes in escapement, recruitment, market characteristics and fishery practices on the ocean fishery. These answers would include estimates of economic surplus, shadow prices, catch levels and composition, numbers of salmon escaping by type, age and time period, subpopulations of salmon remaining in the fishery throughout the focal interval, and the distribution of the salmon population remaining in the fishery at the end of the model period.

Development and maintenance of the proposed model can be compartmentalized. Individuals involved in developing the objective function do not have to also estimate survival rates or develop the population dynamics system. Programmers involved in developing the solution procedure do not have to also formulate the model. Thus, the potential exists for good project management to enhance the development and maintenance of the proposed ocean salmon fishery model.

The proposed model is based on the probability-of-catch assumption as specified earlier. It also involves the assumption that biological growth, mortality, fishing effort and intensity are exogenous components that determine model parameters and variables, rather than output to be estimated during the analysis. For the purposes of the analysis of fishery management policies and practices, however, we believe this type of mathematical programming model has considerable advantages. If a mathematical program is adopted, we recommend serious consideration of the nonlinear ocean salmon fishery model described in this paper.

Chapter 3

Procedures for Valuing Changes in Ocean Salmon Stocks

INTRODUCTION

This chapter describes various approaches for measuring the economic value of harvesting enhanced Columbia River salmon stocks. Such values are necessary to the mathematical programming model reviewed in the last chapter as well as being of interest in their own right.

Assessment of the economic benefits or losses from actions which result from changes in the ocean fisheries should account for the gains and losses to individuals in their roles of consumers, producers and resource suppliers. The notion of economic benefits includes the concepts of producers surplus and rents to resource suppliers, that is the returns over and above the costs of doing business. Consumer surplus is analogous, representing the difference between the maximum amount of money an individual would be willing to pay and that which he must pay in the market to enjoy the use and consumption of a commodity.

These two concepts can be applied equally well to the commercial and recreational salmon fishing sectors. The benefits from increasing salmon available to the commercial sector (or the losses from reduced salmon stocks) will include the change in surpluses to fishermen, processors, retailers, and the ultimate consumers. Of ten, these can be deduced from information on aggregate demand and supply in the salmon market, information typically available as a consequence of market transactions. The specific application of the surplus concepts to the commercial sector and the problems which arise in empirical measurement are discussed in the next section.

The recreational sector presents a greater conceptual and empirical challenge. The benef i t measure (consumer surplus or willingness- to-pay) is applicable to recreationists. However, there is no clearly defined market as in the commercial sector. The nature of the recreational experience further complicates the research. The recreationist is consuming a commodity which is more than just salmon; he is consuming a recreational fishing experience which is enhanced by increased salmon catches. When the commodity in question is not marketed, individuals' surpluses (or willingness-to-pay) cannot be calculated from market demand functions, and standard techniques for approximating this willingness-to-pay measure using market data cannot be employed. See Rahmatian, 1987, for further discussion of components of willingness-to-pay for elements of the environment.

When no markets exist for the commodity the researcher can choose between two approaches. He can identify markets for related goods, making indirect calculations of willingness-to-pay, or he can ask individuals directly what their willingness-to-pay would be. The various approaches, together with their strengths and weaknesses, are described in a later section.

The discussions in following sections are predicated on the assumption that fisheries management decisions are independent of the water allocation schemes. Emphasis is placed on the importance of predicting management strategies and understanding bioeconomic interactions in estimating long-run economic benefits. The final section provides suggested procedures for obtaining estimates of the welfare effects of increased salmon harvests.

BENEFITS OF COMMERCIALLY HARVESTED SALMON

Actions which change salmon stocks affect the well-being of individuals in the commercial sector in a variety of ways. First, for the offshore harvesters, changes in stock density affect the cost per pound at which various quantities of salmon can be harvested. Greater stock abundance reduces the of production costs. Second, reductions in costs at the harvesting level lead to declines in the price of salmon. Greater stock densities, which lead to lower ex-vessel prices and greater volume, can lower average costs of processing and marketing fish and, ultimately lower retail prices. Increased stock densities, therefore, affect harvesters, middlemen (processors and marketers), and consumers. The discussion in subsequent sections explains how the benefits of enhanced salmon stocks can be measured for each group.

The effects on the commercial sector will differ in the short-run from the long-run, and we discuss the implications for both. The short-run analysis is important because observations of fishing operations will typically be made at points of short-run but not necessarily long-run equilibrium. The nature of fishery management decisions, random variation in fish stocks, and other changing conditions in the salmon fishery conspire jointly to prevent a long-run equilibrium from emerging. However, the long-run is of relevance because the producer surplus which is likely to accrue to harvesters in the short-run may not be sustained if new harvesters are drawn into the salmon fishery.

RETURNS TO HARVESTERS

The Short-Run

Economists are agreed upon the correct way to measure benefits accruing to the producing sector of the economy. These benefits (producer surpluses) equal returns to resources such as land, labor, managerial experience, and other resources over and above the returns they could earn in their best alternative use. They include what are usually referred to as rents (including short-run quasi-rents) and pure profits. Producer benefits are conceptually easy to measure, but since they require information on returns as well as potential returns to alternative uses, they are often practically difficult to calculate.

The producers' surplus is the excess of revenue over costs. To the extent that the industry supply curve captures the marginal costs of producing the commodity, the area beneath price and above the supply curve is producer surplus. Thus to measure producer surplus one needs to know the supply curve and the equilibrium price.

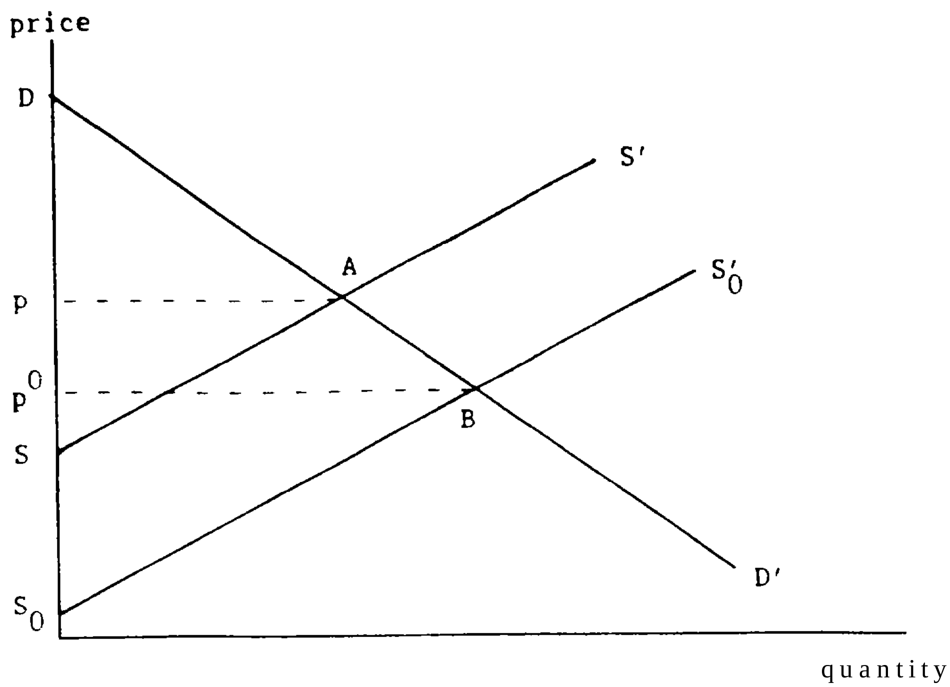
The supply curve for salmon incorporates all costs, out-of-pocket and opportunity, that harvesters incur when they land more fish. It is conditional on the price of gasoline and other inputs, the alternatives available to industry participants and the size of the salmon run. As these factors change, the supply curve changes position resulting in changes in producer surplus.

As availability of salmon stocks grows, the shift outward in supply will alter market price so that the change in producer surplus will be equal to the difference between triangle p^oBS_o and triangle pAS in figure 3.1. Of course knowledge of the demand curve is necessary to determine the new equilibrium price and thus the change in producer surplus. With enough information, one could show how short-run producers' surplus would change with any given level of the salmon runs.

The concept of producers' surplus is clear, but the estimation of it is difficult. The essence of the task is the estimation of short-run supply functions (such as those pictured in figure 3.1) which relate price to harvest levels and are conditioned on input prices, stocks, and the opportunity cost of special resources such as labor. The alternatives available to the fisherman determine the opportunity cost of labor, but these are not in general well understood. Some have argued that if fishermen were not fishing, they would be unemployed, and thus their opportunity cost to society is zero. If this were true and there were no other costs of production, then changes in producer surplus could be approximated by the change in total revenues. Unfortunately, neither assumption is likely to be true.

Nonetheless, the change in revenues associated with an increased catch run could be used as a first approximation to producer surplus changes. This is not as bad an approximation as it might first appear to be, since we are interested in the change in producer surplus associated with a change in salmon stocks, not the total producer surplus. Clearly in the polar case of perfectly inelastic supply, the change in revenues equals the change in

Figure 3.1. Producer Surplus



producer surplus. In less extreme cases, the necessary condition for the change in revenues to approximately equal the change in producer surplus is that total costs change little as stocks are increased. Marginal and average costs fall with increased stocks, but total harvest and therefore total costs increase.

A difficulty arises because of the constraints management agencies have placed on the commercial harvesters. Seasonal restrictions, sometimes as short as three days, can impose severe limits on how much additional fish a vessel can catch, given the period of time allotted. However, the constraints may make the change in producer surplus easier to compute. That is, fishermen will take the same number of trips regardless of stock size given the seasonal constraints. Increased stock size therefore will increase catch and not alter costs substantially. Revenue changes thus would reflect producer surplus changes.

In most cases precise producer surplus measures require detailed information about vessel costs and production and knowledge of management response to increased salmon runs. Information on costs is available for other fisheries in the form of cost and production simulators. A simulation of salmon vessel operations would help determine the extent to which catches would be increased and costs would change with increased runs.

Entry of Firms and Endogenity of Salmon Stocks

While changes in total revenues may reflect producer surplus in the short run, it is doubtful that these, even with refinements, can be used to reflect long-run producers surplus accurately. One reason relates to the long-run entry of fishing firms into the existing fleet.

There is undoubtedly a level of increased salmon stock which will induce entry of either inputs from other fisheries or new inputs from existing firms. As profits rise with increases in catch, existing firms encounter constraints such as the vessel's holding capacity or the size of the harvesting gear. Likewise, increases in catch may suggest to other vessel owners that the Columbia River salmon runs are a better alternative than their existing operations.

In either case, one cannot disregard the opportunity costs of the new investment. Not only does the new investment represent opportunities foregone in other sectors of the economy, but the reassignment of vessels to the Columbia River system will likely reduce production and consumption of fisheries products from other areas. Knowledge of investment and fishery transfer decisions is essential in the long-run analysis.

Entry and investment are probably best addressed in a discrete choice framework. Bockstael and Opaluch (1983) have demonstrated this approach in their study of the New England fisheries. Their results indicate fishing firms change the nature of their fishing operations if incentives are sufficient. To our knowledge, no one has attempted to apply this approach to the Pacific Coast fisheries.

This issue of increased offshore effort raises an important point for long-run analysis. An initial change in the salmon run does not guarantee a change of similar magnitude in the long-run stock. First, the existing fleet may harvest sufficient volume so that the increase is not viable in the long-run. Second, if the existing fleet does not diminish the enhancement in stocks, then there is incentive for more effort to be directed to the Columbia system. The harvest associated with the new effort will likely reduce stock improvements caused by the policy change.

The point is that salmon stock size should be treated as endogenous in a long-run model. The ability to link the economics of harvest and entry with intertemporal biological change may be critical to the successful valuation of salmon stocks. Interdisciplinary research in the long-run dynamics of investment and biological change is the only way to address this issue.

Consumer Benefits from Increased Salmon Runs

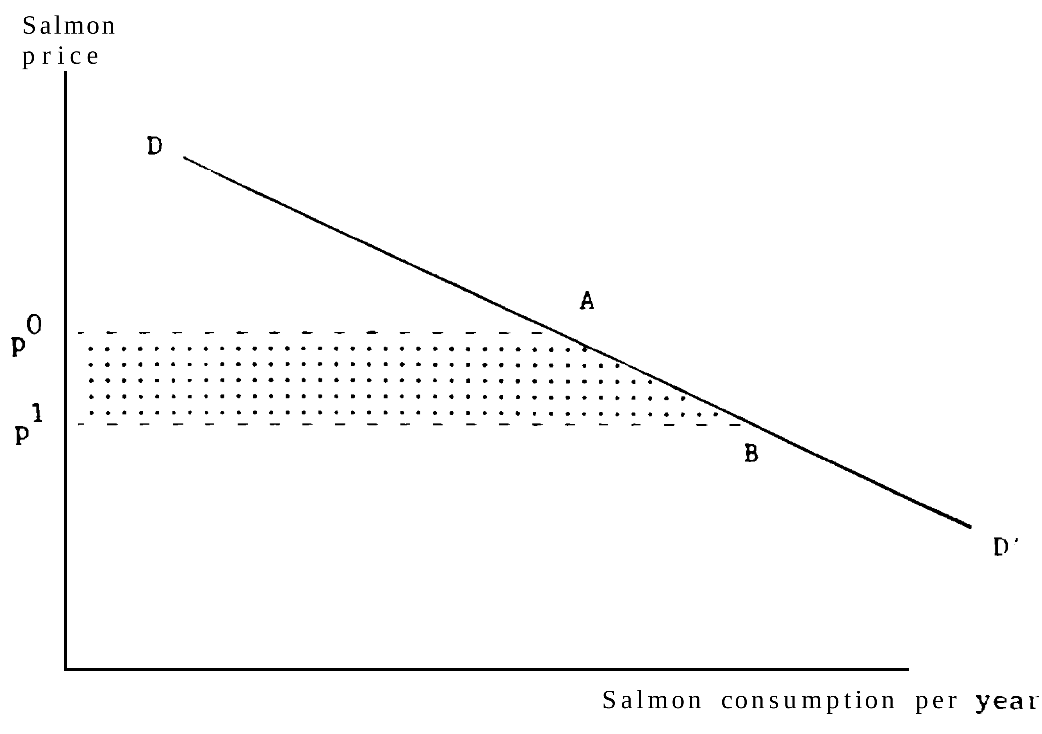
The estimation of consumer benefits is a controversial concept. However, after many years of debate over the issue, a consensus is forming within the economics profession as to practical and defensible measures of the consumer benefits associated with events which alter the price of a commodity. (See Chapter 5 of Just, Hueth and Schmitz, 1982, for a complete development of these arguments). The "willingness-to-pay" for a price change has been accepted as an unambiguous money measure[1] of consumer benefits. This measure is defined as the amount of money which the individual must pay, or with which he must be compensated, such that he is indifferent between his original situation and that following the public policy action which alters market price.

The willingness-to-pay to avoid a price increase or to enjoy a price decrease can be approximated by the change in the area to the left of an individual's demand curve and above price. For example, if an external event were to cause a decline in the price of salmon from some price p^o to p^1, then the trapezoidal area (p^oABp^1) in figure 3.2 would be an approximate measure of the individual's willingness-to-pay for the right to consume his desired amount of the good at the new, lower price. This area is the precise "willingness-to-pay" measure only if the demand curve is a "compensated" demand curve.[2] However, economists (e.g., Willig, 1976 and Rahmatian, 1979) have shown that under many circumstances consumer surplus is a good approximation of the more correct measure. A small income effect is a condition for this approximation to be close, but it is quite possible that the demand for salmon has a rather large income effect, salmon being something of a luxury good. In any event bounds on compensating and equivalent variation can be established from the consumer surplus estimate and information about the size of the income effect.

Welfare Measurement in a Vertical Market

The measurement of consumer surplus requires that we know not only the

Figure 3.2. Consumer Surplus

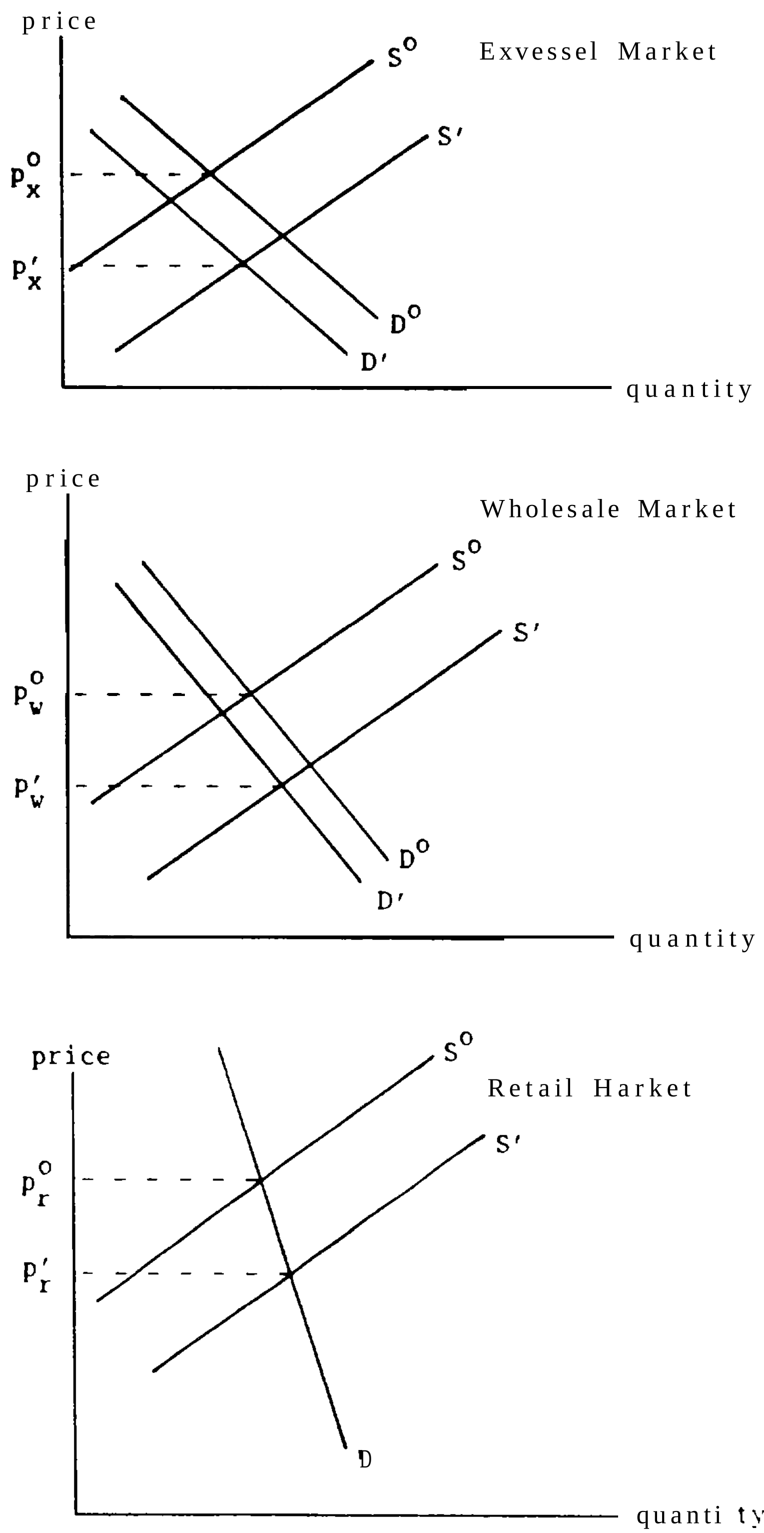


shape of consumer demand but the new equilibrium price. If fishermen sold salmon directly to final consumers, the story told so far would be sufficient to explain all prices and to capture all benefits. Middlemen (i.e. the processors, marketers and restaurant operators) buy salmon to provide a product to the ultimate consumer. These agents are important because they determine how price changes will be passed on to consumers and because welfare gains can accrue to them as well.

When exogenous shocks (such as the effects of increased salmon availability) are introduced in markets, there are repercussions throughout the marketing chain. In the case of salmon, factors which affect catch rates will shift the supply of salmon outward, at least in the short-run. This shift will cause the ex-vessel price of salmon to fall to wholesalers and processors and result in a shift outward in the supply of salmon at the next marketing level because an input price has fallen. This causes the price in the wholesale market to decline. Ultimately, retailers face lower prices for their supplies, which shifts their supply curve outward, lowering the price the consumer faces. Of course there are second round effects as well. Lower wholesale prices shift the ex-vessel demand function to the left which in turn alters price once again, and so on. The series of adjustments eventually results in a new set of equilibrium prices being achieved at each marketing level such as those depicted in figure 3.3. The point is that an exogenous change anywhere in the marketing chain will generate repercussions in the form of price and welfare changes throughout.

The salmon marketing chain is not so simple as that depicted in figure 3.3, but the same principle holds no matter how many levels there are to the marketing chain. In fact, the effects need not even be limited to the vertical marketing chain. If the above price changes cause shifts in demands for other inputs at any stage in the marketing chain and if these shifts cause a change in the price of the other input (i.e., supply is not perfectly elastic), then there will be welfare effects in these horizontally r-elated markets as well.

Figure 3.3. Adjustments to Exogenous Shocks Throughout Marketing Chain



Welfare measurement in interrelated markets is the subject of a 1979 paper by Just and Hueth and it is explored in great depth in Chapters 8 and 9 of Just, Hueth, and Schmitz (1982). In the latter the authors show that there are two ways to measure the full complement of welfare changes resulting from an exogenous disturbance. The first is to measure each market groups' welfare separately. This requires estimating market supply and demand functions for each market level in the system. The second is to obtain aggregate welfare measures across groups by estimating sector supply and demand curves in the market where the exogenous disturbance occurs. A sector demand curve is one which relates quantity demanded in market k to the price in market k and to all exogenous factors affecting buyers in this market and all agents farther up the marketing chain. Likewise a sector supply curve relates quantity supplied in market k to price in market k and to exogenous factors affecting suppliers in this market and all agents farther down the marketing chain. Thus a quasi-reduced form equation results; endogenous prices at other levels of the marketing chain are absent but exogenous factors which affect them are included.

The point can be illustrated in the context of a simple two-level market sector. Consider the market supply and demand functions for each market level (e.g., the primary level (1), and the retail level (2)):

$$S_1 = f^1(p_1, Z) \tag{3-1a}$$

$$D_1 = g^1(p_1, p_2, W) \tag{3-1b}$$

$$S_2 = f^2(p_1, p_2, W) \tag{3-1c}$$

$$D_2 = g^2(p_2, Y) \tag{3-1d}$$

where S_i, D_i, and p_i represent the supply, demand and price of the good in market level i. The vectors Z, W, and Y represent exogenous factors which enter the decision problem of the resource supplier, the middleman, and the final consumer, respectively.

If the system in (1) were estimated simultaneously, welfare measures of an exogenous change could be assessed for each type of economic agent. The parameters of system (1) would be estimated using some simultaneous equation method (two or three stage least squares or maximum likelihood) and the new equilibrium quantities and prices would be predicted from the reduced forms.

The effects of an exogenous shift in the primary market supply function are illustrated in figure 3.4. Once having estimated the functions and determined the new equilibriums, the welfare effects to each group could be determined as follows. Resource suppliers gain F+G-(C+D+F) = G-C-D. Middlemen gain A+C+E-(A+B) = C+E-B, which could alternatively be measured in the output market as f+g-(c+f) = f-c (see Just, Hueth, and Schmitz, 1982). Consumers gain c+e.

An alternative method is to estimate a market supply function for the resource suppliers but a sector demand function in the primary market. The sector demand function is defined as a equilibrium demand function in the sense that it implicitly incorporates all price changes in the higher level markets in the marketing chain. In the above example, the sector demand function in the primary market would take the form:

$$D_1^* = h(p_1, W, Y). \qquad (3\text{-}2)$$

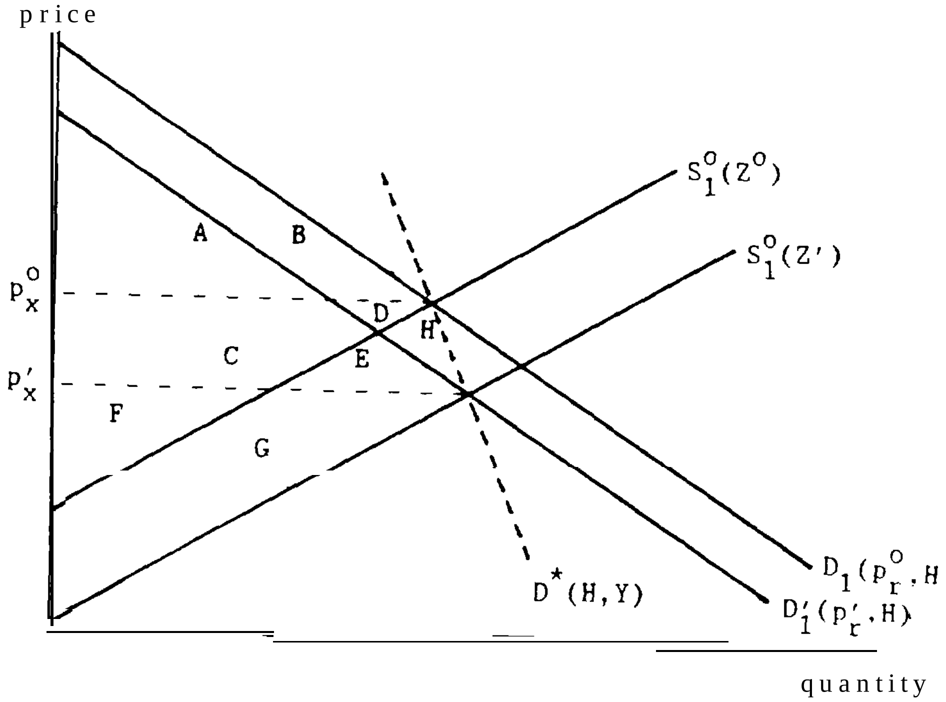
This function incorporates the adjustments in p_2 which would come about with a change in p1 resulting from an exogenous shock in market 1. Formally,

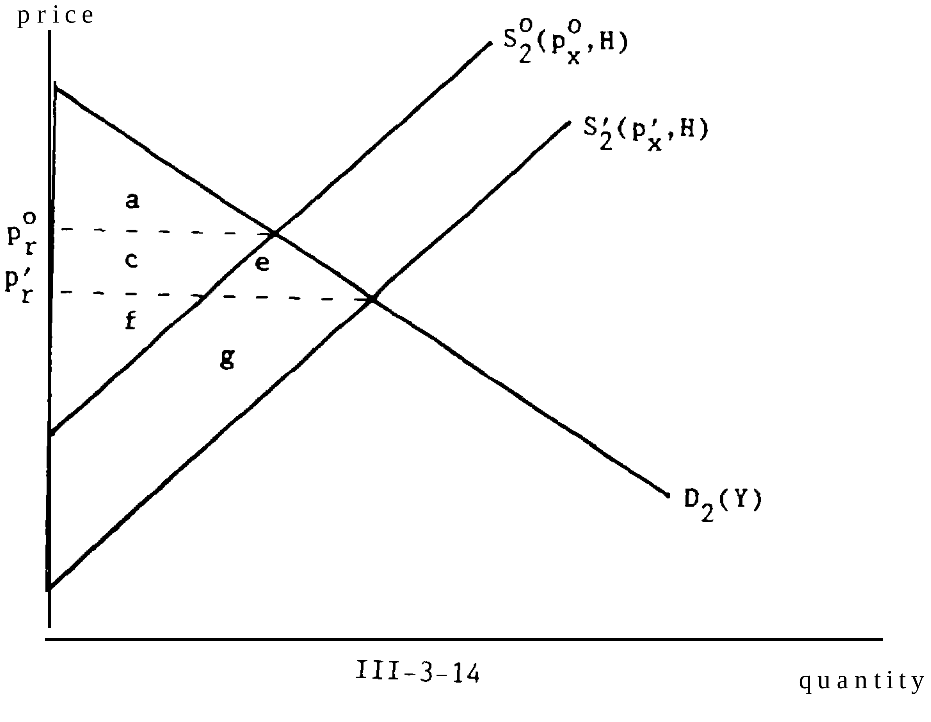
$$D_2 = S_2$$

which implies

$$g^2(p_2, Y) = f^2(p_1, p_2, W).$$

Figure 3.4. Welfare Measures in Vertically Related Markets





Solving for p_2 gives

$$p_2 = k(p_1,W,Y)$$

which can be substituted into (3-1b) to yield the sector demand in (3-2). Intuitively, D_1* traces out the relevant points on a shifting demand function in market 1.

Empirically, there are advantages and disadvantages in estimating the entire system (3-la thru 3-ld) and in estimating the market supply (3-la) and the sector demand (3-2). The major advantage in the system approach lies in the additional knowledge obtained as well as in the potentially more robust welfare estimates. The disadvantage is the likely collinearity among prices in (3-lb) and (3-1c). Sector demand and supply eliminate much of the multi-collinearity problem but yield less information about welfare effects at each individual marketing level. Our inclination would be to estimate the sector demand, however, because the multi-collinearity problems are certain to be severe.

Sector demand functions have been estimated in fisheries (see Bell, 1968) and their usefulness relative to the complete system model frequently debated (Lin and Williams, 1985). What has not been appreciated is the distinction in welfare measures which are obtained from the two estimation approaches. Careful selection of arguments for the sector demand function is critical for meaningful welfare measures. In the salmon problem, careful specification is needed to reflect factors such as excess processing capacity and the nature of the international demand for and supply of salmon.

BENEFITS OF RECREATIONALLY HARVESTED SALMON

While the concept of consumer surplus (or willingness-to-pay) developed in the last section is relevant to the discussion of recreational benefits, the methods used for its estimation are not. Since no well defined market exists for changes in salmon catches by recreationists, the benefits or losses from such changes must be deduced by less direct means.

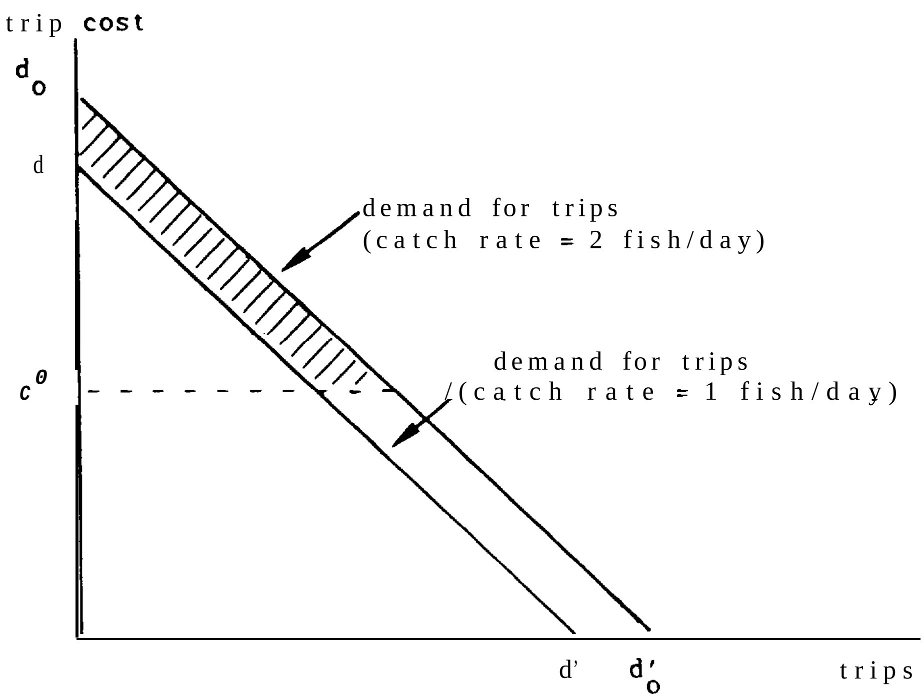
The distinction between short-run and long-run, discussed in the previous section, is relevant to the recreational sector also. For analyzing recreational benefits, fish densities and the number of anglers are viewed as exogenous in the short-run but become endogenous in long-run analysis.

We reasoned in the previous section that producers' surplus depends on salmon stocks. The same reasoning holds for recreational anglers, but the monetary measure of satisfaction is consumer surplus. The value of access to salmon fishing can be measured in terms of consumer surplus. This value of access, and thus the consumer surplus measure, depends in part on the expected catch rate, which in turn depends on salmon stocks.

This relationship can be shown graphically. In figure 3.5, let dd' be the demand for access to the site, given an expected catch rate of say one per trip. An angler with a cost per trip of c^o would be willing to pay the area above the cost curve and below the demand curve dd' for access to salmon fishing. When salmon stocks increase, the expected catch rate increases, and the demand for access to salmon fishing increases. Suppose the expected catch rate increases from one to two. The demand for access increases, and the willingness-to-pay for access becomes the area above the cost line and below the demand curve $d_o d_o'$. The net change in willingness-to-pay is the cross-hatched strip between the two demand curves. This area is what a representative angler would pay to have the salmon stocks increased so that expected catch increased from one to two fish.

This measure of the benefits of changing salmon densities is commonly used and is based on the theory of quality-differentiated goods. For the method to work well, anglers of a given type (age, skill, experience, etc.) must regard expected catch as a fixed characteristic of the fishing trip, much as they would regard the weather or water quality. If the assumption of exogenous catch rate is grossly wrong, the quality-differentiated model, on

Figure 3.5. Valuing Quality Changes



which studies such as Brown (1985) and Loomis (1986) are based, will be invalid. Exogenous quality, as described in Bockstael and McConnell (1981), is a condition required to make the model econometrically feasible. For the change in areas behind curves to be relatively complete measures of the value of change in fish stocks to the angler, two additional assumptions are required. The first is weak complementarity: if the angler does not go fishing, he does not care about changes in salmon densities.[3] The second is that the marginal value of changes in fish stocks be independent of income. This condition, put forth by Willig (1978), makes consumer surplus a good approximation of the more correct variational measure. In its absence, consumer surplus may not only be a poor approximation to the accepted welfare measures, but may fail even to be bounded by compensating and equivalent variation, a condition which always holds for price changes.

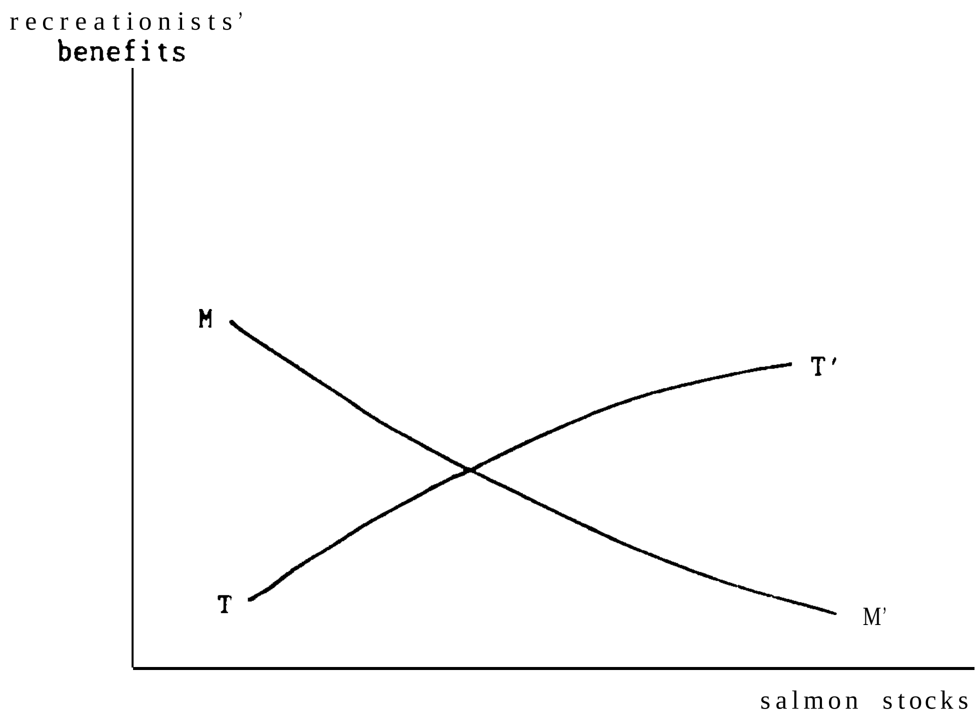
To find the aggregate value of changing salmon stocks, we need to add up cross-hatched areas for all anglers. Further, by observing the effects of additional fish stocks on demand functions we can calculate any such effect on aggregate consumers' surplus. As salmon stocks grow, surplus will tend to increase but at a decreasing rate. Eventually increases in salmon stocks will bring negligible additional increases in anglers' surpluses. The expected relationship is shown in figure 3.6, where TT' represents the total value of salmon stocks, and MM' represents the marginal value. The essence is that the marginal value of salmon stocks depends on the level of salmon stocks and tends to decline with increasing stocks.

There are two prevalent methods for measuring the benefits shown in figure 3.5. The first depends on recreationists' behavior and is developed analogously to the market measures. The second relies on direct questioning of recreation participants.

Market-Oriented Methods

Market-oriented methods for valuing recreation commodities use observations on an individual's behavior in markets for- related goods to

Figure 3.6. Total (T) and Marginal (H) Benefits of Increased Salmon Stocks



infer his valuations for the non-market good. Development of these methods began in the late 1950s (Clawson, 1959; Clawson and Knetsch, 1966). The most widely used technique to emerge from the early work is the "travel-cost method", which examines the relationship between a person's or household's consumption of a non-market good and the cost of travel incurred in order to consume the good. Responses to changes in travel cost are assumed to be similar to responses to changes in price for the non-market good.

More recently, the travel-cost method has been shown to be a special case, in which quality is exogenous, of a broader behavioral model referred to as the "household production" model (Bockstael and McConnell, 1981 and 1983). In the household production model the individual buys market goods and combines them with time in a "production" process to produce non-marketed commodities. A salmon fishing trip, for example, is produced by an angler using time together with purchased inputs such as gasoline, tackle and bait. The result of this production process can be considered the sport fishing experience. The individual's demand for sport fishing trips will be affected by the cost of producing those trips and by the quality of the trips, where the quality dimension may include such factors as the natural beauty of the site, the solitude, and the number of salmon caught.

The remainder of the discussion will focus on valuing changes in exogenous factors which alter the quality of a sport fishing experience. Because we are analyzing the effects of additional salmon stocks, we discuss methods of measuring only one quality dimension, the catch rate associated with a salmon sport fishing trip. As figure 3.5 shows, behavioral measures of willingness-to-pay for salmon stock changes rely on demand curves. The travel cost method is a way to estimate these demands.

The Simple Travel Cost Model

The zonal travel cost model involves construction of a demand function for visits to a site by viewing trip costs per visit as price and visitation rate as quantity. Visitation rates to the site per individual, or per geographically-similar groups of individuals, are anticipated to fall as distance from the site and travel costs (price) rise. In a graph of the demand function for trips, the price which would normally be on the vertical axis is replaced by costs, principally costs of travel, and the quantity (on the horizontal axis) is trips per person per time period (e.g., per year).

More sophisticated versions of the travel cost model use individual observations rather than observations on zones. Recent research (Smith and Desvousges, 1985; Bockstael, Hanemann, and Strand, 1986) has focused on the virtues of using individual observations to handle correctly the individual's decision to participate. The individual angler's demand function is estimated using maximum likelihood procedures which account for the truncated or censored sample problem inherent in individual data where many individuals are non-participants and consume zero quantities of the good.

There is a vast literature (see Bockstael, Hanemann, and Strand) examining travel cost models. Fundamental issues raised in this literature include how the visitors' time should be valued when estimating the relationship, how to aggregate individual demands to determine aggregate willingness-to-pay, and how to consider likely alternative recreation sites when estimating the relationship. Additionally, the sample of observations must be free of sampling design error and appropriate econometric methods must be used to guarantee reliable estimates (Rahmatian, 1987 and Thayer, 1981).

The credibility of the travel cost method has suffered from many specification and estimation problems. One perennial problem is the treatment of the value of time in a convincing and logical manner. By the 1960s, economists were arguing that time as well as money costs were important factors affecting individuals' decisions to make trips to a site. If time requirements were not included in the demand function, biased estimates of both demand function parameters and consumer surplus measures would result . It was also argued that time was valued differently by different people and these differences could perhaps be captured by valuing time in a way related to these individuals' wage rates. With little or no theoretical defense, a number of methods for valuing time appeared in the literature ranging from the inclusion of time costs per se to incorporating time valued at some fraction of the individual's wage rate.

Study after study has reinforced the conclusion that consumer surplus measures are especially sensitive to the treatment of time. In an attempt to remove some of the ad hoc nature of time specifications, McConnell and Strand (1981) developed a technique to estimate the proportion of the wage rate at which time might be valued. Desvousges, Smith and McGivney (1983) attempted a more defensible household production-related treatment of time.

Recent work by Bockstael, Strand, and Hanemann develops a model of the demand for recreation which explicitly incorporates the valuation of time as it depends on an individual's employment opportunities. The model shows how specifications of demand functions vary, depending on an individual's ability to work extra hours for extra wages. With discretionary employment, utility maximization implies the demand for visits to a site is:

$$x = f(p + wt,z)$$

where p is travel cost, t is the travel time per visit, w the after-tax wage rate and z a vector of other exogenous variables including income. When the individual has no discretionary work, the demand function becomes

$$x = f(p,t,z)$$

where the exogenous variables now include a standard measure of income and a measure of total time available for non-work activities. For a particular class of utility function, Bockstael, Strand and Hanemann show that the demand functions are specified as

$$x = \alpha_0 + \alpha_1(p + wt) + \alpha_2 Z + \text{Error}$$

for individuals who can choose to work extra hours. The α's are combinations of parameters of the preference function. For individuals with no such discretion, the demand functions are

$$x = -r_0 + \tau_1 p + \tau_2 t + \tau_3 Z + \text{Error}$$

where the τ's are different combinations of parameters of the same preference function.

A second, much debated issue in the travel cost literature is the problem of proper aggregation of values across individuals. There are many topics which fall under the rubric of "aggregation", and most are currently unresolved. One aggregation issue stems from the nature of the demand for visits to a recreational site. Many individuals are observed to choose zero visits, i.e., they do not participate in the recreation activity. As the price per visit rises, some people take fewer visits, and others stop visiting altogether; of course, those who were non-participants before do not change their behavior. For goods where there are many observations of zero levels of consumption, econometric estimation and sampling design must be handled carefully--and will differ depending on whether observations are collected for individuals or averaged across individuals within a zone. The nature of the error which arises from zonal based models is examined in McConnell and Bockstael (1984). The problems of zonal models have been the focus of considerable research by W. Brown and colleagues (see for example, Brown and Nawas, 1973; Brown, Sorhus, Chou-Yang, and Richards, 1983). Brown has proposed several adjustments to single OLS models to correct for the censoring/truncation problem created by the participation decision. Brown's suggestion pertains to estimating the individual's demand function, given in the form

$$x_{ij} = f(z_{ij})$$

where x_{ij} is the trips for person i, travel zone j, $f(\cdot)$ is the demand function and z_{ij} its vector of arguments, including travel cost. Brown shows in a numerical example that transformations of the dependent variable yield a correct estimate of consumer surplus. The transformed model is

$$x^*_{ij} = f(z_{ij})$$

where $x^*_{ij} = f_{ij} x_{ij}$, f_{ij} being the ratio of the sample expansion factor for the jth zone to the respondent's share of his distance zone's population. Brown uses this formulation to estimate an approximate per capita demand function which gives reasonably accurate results for consumer surplus, not to estimate the correct individual behavioral parameters.

The problem of truncated/censored errors and participation has been approached through classic discrete choice models in McConnell and Bockstael (1984) and Bockstael, Strand and Hanemann (1985). The basic approach is to recognize that individual demands can be written:

$$x_{ij} = \begin{cases} f(z_{ij}) - \varepsilon & \text{if } f(\cdot) - \varepsilon_{ij} > 0 \\ 0 & \text{if } f(\cdot) - \varepsilon_{ij} \leq 0. \end{cases}$$

Using this formulation, we can estimate the parameters inherent in $f(\cdot)$ by maximum likelihood methods. This model has the virtue of formulating the censored/truncated error as a direct consequence of the participation decision. The probability of participation is:

$$\text{Prob } (f(z_{ij}) - \varepsilon > 0) = \text{Prob } (\varepsilon < f(z_{ij})).$$

With ε distributed normally, the probability function can be estimated by profit procedures from a sample of the population, without information on trips. The full behavioral function can be estimated from a sample of users only. These limited dependent variable procedures require software packages such as LIMDEP or SHAZAM.

All of the above problems arise when researchers attempt to value a recreational site. In the quality-differentiated model, site valuation is intrinsic to the valuation of quality changes. Many site valuation issues arise in valuing the changes in the quality of sport fishing which are attributable to changes in catch. However, the valuation of an improvement in catch rates requires carrying the analysis one step further.

Frequently, attempts to value a quality improvement precede the actual implementation of the improvement. In such cases, demand after the quality change cannot be observed. The change in demand must be deduced from whatever information exists about the relationship between demand and quality. The relationship between visits and quality can often be obtained by observing demand across sites with different qualities. If it is possible to observe individuals' behavior over time, this will also provide the necessary variation.

Multiple Site Models

Once it is recognized that the relationship between the demand for a recreational activity and its quality is best estimated by observing demand over alternative sites or situations (e.g. seasons) with differing quality, models must be constructed to incorporate choice among these alternatives. Such models are valuable in accounting for substitutes.

If substitutes are not appropriately modeled, a number of biases can arise in the estimation of site demand functions and consumer surplus. Demand functions based on the implicit assumption that substitution among sites is not possible may over-value sites when substitution is possible. The nature of the single site travel cost method is to extrapolate use rates across individuals with varying travel costs. The problem with treating increasing travel costs as though they were increasing prices is that individuals in different geographical areas may have different substitutes. If these are not taken into account, the wrong inferences will be drawn from the estimated demand functions because an increase in price (travel cost) will result in the user switching to the substitute, not just visiting the site less often.

Early recreational demand models typically did not incorporate quality directly because the models did not allow for variation in quality, over individual observations. Site demand functions could not be estimated as a function of the constant quality dimension. More recent investigations have found ways around this problem by imposing some degree of parallel structure across the demands for different sites. One approach is to presume that the demand function for each site has exactly the same parameters so that observations across sites as well as individuals can be pooled into one estimation. A more general approach is to estimate a system of demands such as Burt and Brewer (1971) or Cicchetti, Fisher and Smith (1976), but in a second stage, explain the difference in demand parameters across sites as functions of differing quality across sites. This varying parameters model has been extensively employed by Smith, et. al. (1983) and Vaughan and Russell (1982). All of these models encounter difficulties in treating substitute site prices and qualities, however.

In contrast to the models based on systems of demands, a second approach can be developed using discrete choice models. In these models, the dependent variable is discrete or categorical. It represents a choice of one from a finite set of alternatives. As applied to recreational decisions, discrete choice models explain the choice of recreational site on any given occasion as a function of the prices and qualities of all available alternatives. The total number of trips taken in a year (including zero as a possibility) must be estimated in a separate procedure but can be structured to include information from the discrete choice problem.

Bockstael, Hanemann, and Strand estimate such a discrete choice model. The choice among sites is given by the generalized extreme value model (an extension of the logit model) for choice of trip destination:

$$P_i = \frac{\exp\,[\Sigma_j \beta_j Z_{ji}/(1-\sigma)]}{\Sigma_k \exp\,[\Sigma_j \beta_j Z_{jk}/(1-\sigma)]}$$

where P_i is the probability of choosing the ith site, $k = 1, \ldots, K$ is the number of sites, Z_{jk} is the jth attribute of the kth site and á and å are the vectors of parameters to be estimated. This model explains the distribution of trips, but not the total number of trips. In a way which is intuitively plausible but not fully consistent with choice theory, one can construct the inclusive price, defined as

$$I = \ln \sum_{k} \exp \left[\sum_{j} \beta_{j} Z_{jk} / (1-\sigma) \right]$$

for each individual to help explain the total number of trips, for example, in a Tobit model.

The combined discrete and continuous choice models can be used both to predict behavioral responses to sport fishing changes and to value these changes in terms of willingness-to-pay measures. The model captures all phases of the recreational decision--whether or not to participate, how many trips to take, and how to allocate those trips among alternative sites or fishing opportunities.

Travel Cost Studies of Angler Success Rates

Steven's (1966) study of fishing in Yaquina Bay, Oregon, was the first attempt to introduce catch rates (or "angler success") into recreational demand analysis. Using daily and weekly information, he determined short-run elasticities of effort (fishing days) to success rate for salmon (.37 to .58) and bottom fish (.09). The model of demand is relatively ad hoc with the response-to-success rate being estimated independent of a Clawson travel cost model of recreational demand. The ultimate intent of the study was to determine how pollution in Yaquina Bay would influence recreational values from sport fishing. No consideration was given to alternative sites, and the study addressed aggregate rather than individual behavior-.

Other studies have used aggregate demand information (e.g. Brown, Sorhus, and Meyer, 1982; Samples and Bishop, n.d.) by first determining the aggregate consumer surplus for different locations of sites. These aggrega te surpluses are then regressed against the catch rate at the various sites. The coefficient associated with the catch rate is interpreted by the authors as some measure of the aggregate value of a marginal change in the average catch rate.

The authors undoubtedly recognize that catch rates can bring new participants to a site as well as increase the activity of existing fishermen. The work does address this aspect of aggregate response, but is far from an integrated model of individual behavior. For example, catch rate is not treated simultaneously with travel cost when estimating effort at a site, and benefit estimates are inconsistent with any realistic multiple site decision process.

McConnell and Strand (1981) studied the individual response of participants to changes in catch rate. The estimating equation in these models includes travel cost and catch rate directly. Estimated elasticities associated with coastal species such as striped bass ranged from less than 0.1 to over 0.2. The major shortcoming of this work lies in sample selection bias and in its inability to examine the influence of catch rate on participation decisions.

Probably the most advanced empirical work in this area is by Vaughan and Russell (1982). Their work uses a three-step behavioral sequence. The sequential decisions are: (1) whether to be a fisherman or not; (2) whether to participate in fishing; and (3) whether to fish for trout, bass, or roughfish. Presumably, participation in trout and bass fishing is a function of the catch of each type of fish. For example, the fisherman's catch of trout has an effect on the participation in bass fishing and the days of bass fishing, as well as trout fishing. The results show both participation and intensity to be responsive to catch rates, but these factors are difficult to relate to estimates gained in previous studies (like elasticities) without information which is unavailable in the study.

Vaughan and Russell are able to determine how aggregate demand changes with changes in success rate, but they rely on an approximation of compensated measures of value with uncompensated curves. The benefits from water quality improvements are assessed as the difference between the existing situation and a new situation in which water quality has improved. The temporal sequence from changes in water quality to increased productivity of fish stocks is not modeled by Vaughan and Russell, making their work essentially static, requiring fish stocks to adjust instantaneously to changes in water quality.

DIRECT QUESTIONING TECHNIQUES

A Description of The Techniques

For situations in which observations on behavior are not possible, travel cost and other market-related techniques may be difficult to estimate. For example, for the introduction of a new species of fish into a lake previously not fished, no observations on behavior would be possible. Direct questioning techniques (frequently referred to as contingent or hypothetical valuation) are not dependent on observed behavior and thus may at times be the only viable approach.

Contingent valuation requires the design of a hypothetical market in which surveyed individuals can hypothetically participate. The researcher formulates a series of questions designed to elicit directly the individual's willingness-to-pay (bid) for a policy change or other external event. Both the event and the willingness-to-pay bid are hypothetical in that individuals are asked to assume that a change would take place; except in experiments by Bishop and colleagues, the bid is never actually collected or paid.

In concept, the technique is intended to reveal the true willingness-to-pay measure; that which would keep individuals at the same level of satisfaction before and after the policy change. As such, it is free of the estimation problems described in previous sections including the difficulty of estimating a response to an exogenous change which has not already occurred. However, direct questioning techniques generate their own difficulties, which are described fully in Cummings, Brookshire and Schultz (1984).

Contingent valuation is appealing because it can presumably be used to value almost anything, since it relies not on modeling observed behavior but on direct questioning. In addition to all of the potential biases associated with the way in which the questions are asked (see Cummings, et al. for a discussion), however, there are three fundamental problems with the approach. The first is that, given the hypothetical nature of the questioning, there is no way to do any hypothesis testing on the results. The approach simply yields valuation answers and the “truth” is never, even after the fact, observed. Consequently, there is no obvious way to use statistical procedures to determine how close the estimates may be to observed behavior or even how certain the estimates are. The second problem with contingent valuation is that it does not do well if the hypothetical change (in price or quality of an activity or resource) would normally prompt a series of adjustments in behavior. A third problem with contingent valuation, which is of particular relevance here, is that it is principally a valuation technique, not a modeling technique. Thus, it is designed to address the sole objective of estimating welfare gains and losses but not that of estimating behavioral relations. How many additional salmon anglers are drawn by greater runs is still a relevant question even if we knew the value per current angler (See Rahmatian, 1981, for additional insights into these issues.)

It would be possible to estimate demand functions from answers to direct questions if the appropriate questions were asked. One could ask “How many trips of a certain type would you take at a cost of $X?" Ultimately, the data would be used to estimate models statistically, such as those used in recreational demand (travel cost) models. They would be subject to all the same drawbacks of the recreational demand models, but additionally would be based on hypothetical responses. While the hypothetical approach has obvious disadvantages, it does have the advantage of yielding observations on any circumstances one might wish to evaluate. Bishop and Heberlein (1979) went beyond the hypothetical question concept and actually purchased hunting rights from respondents. This approach, however, may not be practical when large populations of respondents are required.

Applications of Hypothetical Valuation

An initial attempt at estimating the value of a catch rate change via contingent valuation is offered by Mathews and Brown (1970). In their study of Washington salmon fisheries, a contingent valuation question was asked regarding the amount of money an individual would be willing to accept in payment for relinquishing the right to fish in each of four regions. These willingness-to-sell bids are then divided by trips to get an "average value per trip" for each of the four regions. Although not analyzed statistically, the graph offers strong evidence of a positive relationship between catch rates and average consumer surplus. The relationship is then used to estimate the effect of lowered catch rates on the consumers' value of sport fishing.

A number of other studies have asked hypothetical questions to determine the value of a fishing trip (e.g. Wegge, Hanemann, and Strand, 1985). In one instance, each individual is asked (directly or through a bidding process) what travel cost would force him to stop taking trips to a site. This information, along with the fisherman's current number of trips and travel cost, gives two points on a demand curve. By assuming a functional form for the demand curve and integrating over the actual number of trips, we can derive value of willingness-to-pay. This value can be regressed against the average catch per trip to determine the marginal willingness-to-pay. In essence, the procedure is similar to the approach by Mathews and Brown, except that willingness-to-pay is estimated.

Long-Run Analysis

The discussion presented above takes a short-run view: fish stocks and anglers are assumed fixed. Long-run developments in a recreational fisher-y.

as in a commercial fishery, make the number of participants and the salmon stocks endogenous. This endogenity can be understood by analyzing the long-run impact of an enhancement program. Suppose such a program has the initial impact of increasing salmon stock densities. Greater densities lead to higher consumer surplus for anglers already fishing for salmon. The increased surplus may attract additional anglers, who in turn harvest more salmon, reducing current salmon stocks. Because salmon stocks are endogenous in the long-run, we must focus on the impact and net benefits of rules and regulations which influence angler behavior and salmon stocks. The decision to participate in salmon angling should be given special attention for long-run analysis.

In planning research on recreational harvesting of salmon, one must think carefully through the implications of endogenous participation and endogenous stocks. First, in the long-run the angler's catch rate, whether per year or per season, becomes endogenous. Rules which enhance salmon stocks, even in the long-run, may increase or reduce the catch rate, depending on the relative responses of angler effort and salmon stocks. Second, the effect of many rules and regulations enter angler's preference or demand functions directly. For example, a daily bag limit has a short-run impact on the value of fishing for an individual angler, and a long-run impact through the salmon stocks.

Rules and regulations in the salmon fishery stem from the inefficient nature of open access to fish stocks. Partial measures to prevent open access have been developed and implemented in the ocean fisheries. The implementation of these schemes may have some impact on the valuation of actions which change fish stock densities. While there are many different actions aimed at curtailing recreational catch, we focus here on the bag limit.

Bag limits which differ by season and location currently apply to ocean salmon fishing. Their effect on recreational behavior depends on how constraining they are and how they influence alternatives. If the bag limit on salmon has the effect of altering the quality of other alternatives (e.g. there are species inter-relationships), then these are missed in contingent valuation. A natural extension of this idea is that contingent valuation questions, once asked, answer one question and cannot subsequently be used to analyze another hypothetical question, i.e., another type of exogenous change in the decision making environment.

If we attempt to use recreational demand modeling in the salmon problem, we first must discover how bag limits affect behavior. If fishermen knew before each trip how much they would catch, and bag limits deterministically altered that catch, then deducing the effects of bag limits would be relatively straightforward. Instead, fishermen probably make decisions to go sport fishing on the basis of, among other things, “priors” on the distribution of fish catch, however vague these may be. Bag limits do not deterministically alter fish catch, they change the probability distribution of fish catch. They truncate the distribution at the bag limit. It is, of course, still possible for a fisherman to go home with fewer fish than the bag limit.

While this introduces some difficulties, it does not make the task impossible. There are several ways of handling the problem. Recreational demand models depend upon modeling recreational decisions as functions of the costs and quality characteristics of all the alternatives. Therefore, it is necessary to get some notion of how the fisherman perceives the distribution of fish catch for all his relevant fishing alternatives. One way to get this information is to ask a series of questions about expectations e.g., “If you went out on ten 8-hour salmon fishing trips, on how many of these trips do you think you would catch fewer than one salmon? fewer than two salmon?. . . " Such questions reveal enough about the individual’s perception of the distribution of the catch to be useful to us, particularly if one of the reference points is the actual bag limit.

Another alternative is to collect data on actual catch rates. This presumes fishermen have perfect information. Then, ascribe to their alternatives the calculated means and variances of the actual catch data. This is much easier but less desirable, since behavior is predicated on perceptions and these could vary a great deal across all sport fishermen. In fact, one of the problems with this approach is that it produces very little variation in the explanatory variables when means and variances of catch rates are calculated from actual data. Moreover, a catch quota will force all of the observations to be two fish or fewer, even though some fishermen may catch more and keep only two. One way to introduce more variation in the data is to calculate means and variances conditioned on the experience of the individual sport fishermen. Note that this approach requires collecting actual catch data by sport fishermen at the end of the trip. (It would be interesting to collect expectations data in pre-trip interviews and actual catch data in post-trip interviews and compare these two approaches.)

An important aspect of bag limits is whether they are enforceable. Do anglers believe they will get caught, convicted and prosecuted because they violate bag limits? If the answer is no, then it is not likely that bag limits will have much influence on the demand for or valuation of the recreational fishery. Further, if bag limits exceed what most anglers expect to catch, then they may have a negligible effect on demand. Ultimately understanding the effect that bag limits have on valuation involves understanding anglers' decision making behavior under uncertainty. These issues are complex and are related to the concept of option value, the persistent study of which has yielded no simple answers.

SUGGESTED PROCEDURES

The Commercial Sector

The calculation of benefits from the commercial sector should include changes in producer surplus to all economic agents in the marketing chain and changes in consumer surplus to final consumers. As described above, these measures can be obtained by estimating appropriate supply and demand functions at the ex-vessel market level. This procedure requires the estimation of a sector demand curve which includes as arguments the ex-vessel price and the important exogenous factors which affect producers and consumers higher up the marketing chain. If such a curve is successfully estimated, then the change in the consumer surplus triangle to the left of this sector demand curve will capture the sum of the welfare effects of all middlemen, retailers and consumers. The condition for this to be true is that the salmon supplied in the ex-vessel market be an essential input in the production of the salmon product which is passed up the marketing chain.

The estimation of ex-vessel supply is likely to be more difficult because of insufficient information about vessel costs and the effect an increase in salmon runs would have on the supply schedule. For this reason, we emphasize the importance of some detailed work of the sort which has been undertaken in the New England groundfish industry and the Gulf shrimp fishery where budget simulators have been developed. Since all the benefits accruing to the commercial sector from enhanced salmon runs will be induced by shifts in the ex-vessel supply curve, the relationship between stock size and supply is critical.

The following outline describes the tasks which need to be accomplished to obtain commercial sector benefits:

1. Develop sufficient information on the cost structure of offshore salmon fishing firms and the constraints which these firms face so that it can be determined whether a significant relationship is likely to exist between the quantity supplied on the one hand and price and size of salmon runs on the other.
2. Identify exogenous factors which affect middlemen, retailers and final consumers in their demand for salmon.
3. Simultaneously estimate the ex-vessel supply and sector demand functions.
4. Determine from hypothetical shifts of the supply function the resulting welfare changes for fishermen and the remainder of the marketing chain.

The Recreational Sector

There are two potentially useful methods for determining the benefits or losses to recreationists from changes in salmon stocks. The strength of the market-oriented methods lies in their ability to predict how people will behave based on how they actually do behave. Researchers thus do not have to rely on "hypothetical" responses but can use real-life situations where factors such as time and income may actually constrain the individual's responses. The general travel cost approach uses analogy for prediction. That is, how individuals behaved in one situation can be used to predict how they will behave in another. Once behavioral models are estimated, welfare measures associated with price or quality changes can be derived.

Hypothetical valuation is useful in that the "analogy" approach can be avoided (particularly when close analogies do not exist). Questions can be posed to an individual which exactly specify the changes which are expected. The individual can therefore give hypothetical answers in response to descriptions of proposed changes. Unfortunately, their valuations will be hypothetical as well and can diverge from the valuations implied by their actual behavior, should the hypothetical situation become an actuality. The literature on the degree of divergence is mixed. No "conventional wisdom" has emerged on whether hypothetical valuation is a legitimate method for estimating benefits.

We think that the behavioral approach is the most fruitful way to estimate the demand for access to salmon fishing. While the behavioral approach may have only slightly better acceptance among researchers doing benefit assessment, it has several advantages. First, the behavioral method is more similar to methods used elsewhere in economics. Economists having no interest in calculating consumer surplus are nonetheless well acquainted with estimating demand functions. Second, the behavioral approach is consistent with the way economists reason. Knowing the behavioral links allows assessment of the results' plausibility. And as our review has shown, there is considerable experience with behavior-based models. This experience pertains not only to the estimation of models, but also to the design of samples.

Behavioral models need a focus of effort. There are several basic relationships which need to be estimated for the models of salmon angling to fulfill their role in measuring the effect of management actions and salmon stock densities. In addition to the several problems of aggregation, valuation of time, and site substitution, the research approach should handle the effects of management actions such as bag limits and stock densities on participation and the level of trips by individual anglers. While the final details of model construction and sample design depend on a considerable knowledge of the details of ocean salmon fishing around the Columbia River, the following plan may be a logical starting point.

1) The recreationist is modeled as choosing to fish among different sites and different parts of the salmon run. The multiple season/multiple site specification allows catch rates to vary and permits the estimation of the effect of catch rates on the demand for trips. The precise nature of the model will depend on the configuration of the data (how many sites and seasons are reasonable alternatives for the recreationists to choose among).

2) The most appropriate sample design is a two-part sample:

   a) field or intercept survey. This part would gather information on trip levels, travel, fee and equipment costs, expected and actual catch, travel time, and other information about individual anglers. The sampling unit should be the representative angler.

   b) a random phone survey of the general population: This part would gather information about participation decisions, and would permit the estimation of the number of anglers as a function of salmon densities. The sampling unit would be the representative household.

3) This sampling plan should be implemented with the punch card system firmly in mind. Both cross-sectional and time-series data from the

punch cards can be used in estimating behavioral responses to increased catches.

The Long-Run

The above procedures will provide good estimates of short run benefits, but these cannot be extrapolated into the future without knowledge of (1) the dynamic bioeconomic relationships and (2) fisheries management policies. Increased salmon runs will induce changes in the behavior of both recreational and commercial fishermen which will have implications for the long-run levels of stocks. Of major interest is whether new commercial fishermen and recreationists will be induced to enter the fishery. Of course, management regimes can alter those implications and it is critical that assumptions about management policies be made explicit when estimating long-run welfare effects.

While dynamic bioeconomic models have been applied to fishery management problems, they have been absent from welfare analysis. Treatment of the long-run welfare implications, taking account of stock dynamics, would be a valuable and pathbreaking contribution.

ENDNOTES

1. The measure is unambiguous in the sense that there always exists a unique answer no matter what pattern of price changes is proposed. However, there are really two relevant measures here: "willingness-to-pay" and "willingness-to-accept payment", and these are expected to differ in most circumstances. Which measure is "appropriate" depends on the type of policy action contemplated and on whether the individual has the "right" to his initial level of satisfaction or to that which he would obtain after the policy change.
2. Compensated and ordinary demand curves differ in the way in which they are constructed. An ordinary demand curve reflects the quantities an individual would choose to purchase at various prices, assuming his income and other prices remain constant. A compensated demand curve reflects quantity demanded at each price given that the individual's income is "adjusted" so that, as prices change, he remains at the same level of utility.
3. This, of course, ignores such concepts as existence value. See Rahmatian, 1979, for example.

Chapter 4

Determination of Economic Impacts

INTRODUCTION

Up until this point, the benefits associated with decisions related to Columbia River fishery stocks have been discussed at a highly aggregated level. Analytical approaches to estimating these aggregate benefits, such as the techniques presented in the previous chapter, have attempted to assess the sum total of the direct economic benefits that various decisions and actions imply, regardless of how the benefits and costs are distributed. However, such approaches may paint only partial pictures of the economic impacts of a decision. Numerous secondary benefits may be realized by activities which either provide inputs to or use the outputs of the commercial or recreational fishing sectors. For example, a large increase in the number of recreational fishermen that visit a harbor town may increase the sales of the restaurants in that town. Furthermore, an aggregate approach usually fails to investigate the income and employment impacts that states, local communities, or specific groups of individuals may experience. Decisions which may appear to yield positive economic benefits on an aggregated national level, for example, could affect specific local communities or groups negatively, and vice-versa.

Researchers have developed techniques to address these secondary and disaggregated impacts, in an attempt to present more complete economic analyses. However, such analyses of the secondary and disaggregated impacts (often referred to collectively as regional economic impacts) provide no indication of the desirability of the changes. Rather, the decision makers must determine what is beneficial and what is undesirable. The impact analysis for example, may show how, as a result of reducing salmon harvest, other industries, regional income and the labor force would be affected. There is no criterion however, such as a benefit-cost ratio, which establishes whether the policy should be adopted according to economic efficiency or other goals. Instead, the analysis merely describes the regional effects of the change.

Nonetheless, it appears desirable to include regional impacts in this Phase II study of ocean fisheries management. Local communities or regions which bear a disproportionate share of the costs or obtain a disproportionate share of the gains clearly may be vocal in their opposition to or support of a particular decision in regard to fishery stocks. A decision which appears to be economically efficient in a broad context may impact a region so severely as to make the decision politically infeasible or questionable on equity grounds. While regional impact studies cannot substitute for decision-making, they can serve with the models in the previous chapter as a tool to help in the evaluation of alternatives.

GENERAL MODEL STRUCTURE AND ALTERNATIVE APPROACHES

The foundation of regional economic impact studies involves analyses that relate output of industries in the area or region in question with inputs needed to produce the output. The use of this information permits the determination of how all outputs and inputs will be effected by a change in one output. The total effect of the change after the adjustments take place can be determined for such measures as value of output, income level, and labor requirements.

The process for determining regional economic impacts is well established. Measurement of the impacts can be made through economic base methods, input-output models, and the adaptation of input-output models to include econometric functions (Proctor; Scott; Nelson and Bender). Scott has provided an excellent description of the differences among the general types of analyses that might be carried out when measuring regional economic impacts, although as Nelson and Bender note the advantages and disadvantages of the alternative approaches are not well understood.

The economic base method describes the overall economy at one point in time. This method assumes that the pattern of purchases, productivity, and all other relationships among industry sectors remains the same over the period of analysis, whether economic activity is increasing or decreasing. For these reasons the economic base method is best applied to small changes in an economy with a relatively stable economic base mix.

Input-output models in their broadest sense show the mix of factors needed to produce various goods, for a given level of technology and fixed prices. They have a high level of industrial disaggregation with linking coefficients that specify the amount one industrial sector buys from or sells to each other sector at some given point in time. Generally, these coefficients are expressed as the amount purchased from or sold to another sector per dollar of output. The advantage of input-output models lies in their ability to trace changes in the activities of a sector that are caused by some initial change in any one of the other sectors to which it is linked. Economies with highly interdependent sectors will produce large changes in total regional output as the economic base changes.

Despite their promise, input-output model have several weaknesses which may limit their usefulness in a study of ocean fisheries management. First, they require a large amount of data to construct. For regions such as the Columbia River Basin, much of this needed data has not been collected, so expensive primary-data surveys may be required to collect enough information to build an input-output model. One possible way to avoid such an expensive survey may be to use readily available secondary data and techniques to massage this data, but this obviously can introduce problems of poor-quality data.

A second, serious potential limitation is that input-output relationships may change over time as the economy of a region changes due to technological advances or other reasons. However, the coefficients of these simplified models are usually not altered during an analysis. Therefore, although the models may be well-suited for short-run analyses when small increments in final demand are being considered, they may encounter difficulties with long-run analyses. Unfortunately, it is the long-run which is relevant to salmon fisheries management decisions.

Third, most basic input-output models treat personal consumer expenditures, local government activities, investment, and exports as exogenous. In reality, however, many of these variable are related to the earnings generated by the industrial output. Furthermore, investment cannot be predetermined because it is related to output through the biology of the stocks and the yield-effort functions. Additionally, the reason exports such as salmon exist is because of the existence of industry output. In short, many of the exogenous variables are directly determined by the sectors represented in the input-output model.

More extensive models can make personal consumer expenditures and local government expenditures endogenous in input-output models by augmenting the inter-industry matrix to include these expenditures as an extra column and adding a corresponding additional income row. However, this procedure encounters the aforementioned problem of fixed coefficients and, most importantly, a fixed relationship between imported goods and locally produced goods. It is not realistic to expect the inputs to remain in fixed proportions over time. In fact, if a new industry locates in a region, a change will take place in all the regional input coefficients that define the relationships among local industries. Technical input coefficients may remain fairly stable for short time periods but for long term forecasting it is necessary to allow them to change. Trade coefficients are even less stable over time, since trade flows are a consequence of supply, demand and transportation costs.

A combined econometric/input-output model could attack some of these limitations and yield improved estimates of the changes in the value added to a region. (See Scott for additional discussion of the advantages of the econometric approach). For example, suppose management decisions caused the salmon fishermen to become unemployed permanently. Income generated in the region would fall, creating a chain of events which would decrease economic activity in all other sectors of the economy. An econometric model could produce estimates of the decrease in fishing employment and income, and the input-output model estimates of the results from the chain of events. An alternative scenario is that the initial unemployment of salmon fishermen would generate a new industry (i.e., non-salmon fishing enterprises) or that the unemployment would be distributed among the other sectors in the region. Again, an econometric approach could estimate the fishing unemployment and predict the employment that each new or established industry could pick up, as well as assess the structural shifts that may take place in the fishing and non-fishing sectors. The input-output model could examine the secondary impacts of this altered economic structure.

ESTIMATING THE SECONDARY IMPACTS IN THE COLUMBIA RIVER BASIN

A number of authors have suggested that the most fruitful approach for estimating the secondary impacts in situations similar to the one described in this research effort may lie in the mixed econometric/input-output technique described above. In particular, the combined use of input-output tables, which have been regionalized with a program such as IMPLAN (U.S. Forest Service) or INFORM (Almon, University of Maryland), and localized analysis may utilize the strengths and sidestep some of the limitations of input-output models. (Harris and Norton; Radtke and Jensen)

In a study that is relevant to the ocean fisheries work here, Norton, Strand and Smith applied the combined Almon national model and the Harris multi-regional, multi-industry forecasting model and supplemented these with substantial county-by-county information gathered through personal interviews in the coastal counties from Maine to South Carolina. The principle driving force in the Harris model is a set of industry location equations that explain changes in output by region (such as by county). The explanatory variables are:

1) partial measures of location rent, which account for the marginal transportation costs of shipping goods out of each region
2) the marginal transportation cost of obtaining inputs at the place of production
3) the cost of labor
4) the value of land or the right to use the resources
5) prior investment
6) prior production
7) demand
8) input scarcity

Once industry output by region is determined, related variables such as investment, employment and earnings can be estimated. The model forecasts equipment purchases by industry and construction expenditures by type of construction. The model also includes a set of socio-economic characteristics that determine population shifts, personal income, and personal consumption expenditures. These reflect the location decisions of individuals.

A study by Radtke and Jensen on the economic impacts of West Coast fishing activities also has used this regionalized econometric/input-output approach. They report:

> The model developed for this assessment study utilizes one of the best known secondary input/output models available. The U.S. Forest Service has developed a computer program called IMPLAN which can be used to construct county or multi-county I/O models for any region in the U.S. The regional I/O models used by the Forest Service are derived from technical coefficients of a national I/O model and localized estimates of total gross outputs by sectors. The computer program (IMPLAN) adjusts the national level data to fit the economic composition and estimated trade balance of a chosen region.

The IMPLAN data base consists of two major parts: 1) estimates of final demand, final payments, gross output and employment for 466 industrial sectors; and 2) a national-level technology matrix. . . . Compared to the development of survey data models, the IMPLAN system is very inexpensive to use.

The IMPLAN model's fish harvesting and fish processing sector was developed for direct use in this application. A major problem with the fish harvesting sector is that many of the fishermen are self-employed and therefore not counted in the labor statistics. The fish processing sector is defined by national processing Standard Industrial Code (SIC) classifications which may be out-of-date and which do not fit the small coastal processor very well.

The solution, in this case, is to accept the IMPLAN model as an analytical description of a local economy and to custom design the purchase requirements for any seafood industry being investigated. This allows for specific assumptions concerning the industry to be introduced into the model. The resulting coefficients are then used to estimate the impacts of fishing industry activities on the local economy.

The I/O model therefore becomes a tool whereby the local industry and local conditions can be simulated. Specific revenues, expenditures, or behavioral assumptions can be changed as conditions change. The resulting coefficients will change as will total estimated impact on the economy.

This is substantially different from the usual approach whereby certain 'coefficients' or 'multipliers' are presented for certain industries without customizing the data for key sectors to fit local conditions or changing behavioral assumptions.

The authors also provide an excellent discussion on the necessity of clarifying the reported multipliers. They conclude that the income coefficient is the most useful among the various multipliers. However, it probably is important to report all multipliers, since this provides the opportunity for users of the research results to examine and compare the alternative measures of generated economic impacts. Scott, for example, points out the necessity of carefully reporting the multipliers and indicates that the meaning of the multiplier depends on whether an economic base, an input-output, or an econometric model is used. He states:

> In summary, the economic ‘multiplier’ derived in a particular study depends on: 1) the type of economic change, size of the change and duration of the change; 2) the type and size of the economy for which the multiplier is calculated; 3) the type of economic effects subsumed in the multiplier (e.g., inter-industry purchases, investment effects); and 4) the period of time over which the economic impacts are allowed to work themselves out. The ‘multiplier’ or prediction of secondary economic impact has no meaning unless the context is specified to incorporate the four dimensions described above. Managers should be very cautious in interpreting or applying any analysis of secondary economic impact which does not specify this context.

A final consideration in the proposed secondary impact analysis centers around the distributional consequences of any management decision. It is important to know which groups (e.g. income groups) bear the brunt of the costs of a particular policy and which groups benefit. Low-income groups, for example, may be more susceptible to some types of economic impacts resulting from particular management decisions. In some cases, they may make up a large share of a work force that potentially will be displaced, be less able to find alternative employment, and have a smaller cushion against economic hardship.

Rose, et. al., have added substantially to the literature on the general subject of measuring the distributional effects of natural resource policies. Their study of the impacts associated with a decision to increase mining activities traces through the potential employment, income, and distributional effects of implementing such a decision, and analyzes how such potential effects may translate into political pressure on decision-makers. Such an approach seems warranted in a study of the impacts associated with ocean fishery management decisions in the Pacific Northwest, and should be included in such a study.

ADDENDUM I

An Approach to Phase III Studies

The modeling approach and the related methodologies discussed in this volume are designed to provide the framework for a comprehensive economic study of the Pacific Northwest ocean salmon fisheries. Results of the analysis can be presented in a variety of ways. Most salient for purposes of the simulation and cost effectiveness modeling, discussed in Volume II, is the opportunity cost of more effective ocean fishery regulation as a mitigation alternative for increasing upstream runs.

There have been many economic studies of the Pacific Northwest salmon fisheries that treat with portions of the research proposed here. These include studies of demand for commercial products, demand for recreational fish, vessel costs and earnings, economic impact and numerous policy issues. Most of these are conceptually and empirically sound. Unfortunately, however, these studies have been somewhat ad hoc and uncoordinated in nature. While they have been useful in addressing specific issues, they have yielded little in terms of overall policy analysis regarding the ocean fisheries along the coasts of the states and British Columbia. This may reflect what seems to be reluctance on the part of agencies to fund large-scale research projects in the area of fisheries economics. These agencies, however, have spent, over the years, hundreds of thousands and perhaps millions of dollars on small projects.

An example of the type of coordination that may be useful is a study that was conducted to address certain policy issues regarding striped bass. In this instance, the Agricultural Experiment Stations and the Sea Grant programs of the University of Connecticut, Cornell University, University of Maryland, and North Carolina State University cooperated with the U.S. Department of the Interior, Fish and Wildlife Service, U.S. Department of Commerce, National Marine Fisheries, the National Sea Grant Office and several state fisheries agencies. These groups jointly funded and carried out a region-wide economics study of the implications of alternative management decisions on the values and impacts, and the distribution of these values and impacts among states.

Economic studies of the Columbia-based ocean fisheries could be approached in a similar manner. This should include the universities, federal and state agencies, the Pacific Fisheries Management Council, and if possible, Canadian universities and agencies.

To successfully establish and coordinate such a study would be a task of major proportions. It would necessarily be a lengthy and expensive process. Core funding by an agency such as BPA would probably be a major factor permitting the success of such an enterprise.

ADDENDUM II

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